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Chapter 7

Conventional Media Filtration with Biological Activities

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Additional information is available at the end of the chapter

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1. Introduction

Conventional gravity filtration takes advantage of gravity of water as a driving force, and is classified as slow media filtration or rapid media filtration. A slow sand filter is simple in design, construction, and operation. It is simply a filter box (usually made of concrete) containing sand media supported by a layer of gravel with appurtenances to deliver and remove water. The first recorded use of slow sand filters for a citywide water supply was in 1804 by John Gibbs in Paisley, Scotland (Barrett et al. 1991). Slow sand filters as their name implies, is accomplished with a relatively slow speed of filtration (typically 0.1 to 0.2 meters per hour) with 1 to 2 meters media depth. Because of the slow filtration rate, the head loss buildup is gradual and usually takes several months to achieve a significant level and form a condensed layer called *schmutzdecke* on media surface, which will be removed manually with media replenishment. Therefore the filter runtime is usually in the magnitude of months as opposed to 24-48 hours with rapid sand filters. A rapid sand filter is operated in a much higher speed (typically 2 to 10 meters per hour) with periodically backwashing the filter to recover headloss which builds up much faster due to a higher filtration speed. Backwashing is initiated normally by set time intervals, headloss across a filter media bed, or filter effluent turbidity. For both slow and rapid filters, filter run times are highly dependent on the freeboard on the top of the media. The freeboard is 1-3 meters, designed according to water qualities, especially turbidity and total suspended solids.

Sand, anthracite, and granular activated carbon or their combination was used as media with proper gradation. Some proprietary media as pumice, expanded clay, diatomaceous earth, and ceramic have also been applied. Characteristics of different media are shown in Table 1.

When media becomes clogged and dirty, the best way is to backwash the filter and flush dirt out. Backwash is classified as fluidized backwash and sub-fluidized backwash. Fluidized backwash requires a higher water rate to expand media bed usually by 20-30%, where the minimal fluidization water velocity is directly related to media type, media size, uniformity

coefficient, water temperature, and salinity (important factor for sea water filtration) (Logsdon et al. 2002). However, a fluidization test is always recommended for precisely identifying the backwash rate to achieve the desired fluidization. Usually sub-fluidized backwash is applied to coarse media with 15-20 m/h, where media will only move or rotate locally and expand slightly as fluidization of larger media require an extremely high water velocity (Logsdon et al. 2002). Backwash can be water only backwash or air assisted backwash, the later of which has gained popularity because of water conservation and effectiveness of media cleaning. The air scouring (usually at 70-90 m/h) can be before water wash, after water wash, simultaneously with water, or combination thereof. Logsdon et al. (2000) summarized the typical water and air rates for backwashing filters with different types of media.

With concurrent air scouring and water wash, the filter bed undergoes a "collapse pulsing action" under optimal air and water rates, which can be predicted according to a set of empirical equations applicable to different media gradation and water temperature (Amirtharajah 1993). Obtaining and sustaining collapse pulse action within the backwashing process is optimum for the removal of particles from the media grains. The collapse pulsing action can be described as follows: the air bubble exits the air delivery device (orifice) and expands under the weight of the media. When the air bubble expands, the media expands slightly within the vicinity of the bubble, and the bubble collapses and reforms just above its original location. This collapsing is due to the weight of the media. The bubble reforms above its original location because the media is only partially expanded. Just prior to collapsing, high local water velocities occur at the perimeter of the bubble. Simultaneous to bubble collapse, media particles rush together and collide in a violent scouring action. This creates a "pulsation" in the bed. The bubble travels on upward, expands, collapses, and re-forms again, and repeats the process several times as it passes through the bed.

Media	Density (g/cm^3)	Major Constituents	Specific Surface Area (SSA) and its References	
			SSA (BET) (m^2/g)	References
Garnet	3.6-4.2	Nesosilicates		(Logsdon et al. 2002)
Sand	2.6	SiO_2	0.04 ±0.001 (0.25 and 0.5 mm size)	(Jerez et al. 2006)
Anthracite	1.6	Carbon	6-7 (0.2-0.4 mm size)	(Davidson et al. 1996)
GAC	1.3-1.5	Carbon	720 (<1 mm size) 900 (0.149 mm size) 928 (0.15 and 0.25 mm size)	(Gergova et al. 1993) (Oliveira et al. 2002) (Tang et al. 2004)
Diatomaceous earth	1.0-1.6	SiO_2, Al_2O_3, Fe_2O_3	27.8 (106–250 mm size)	(Al-Ghouti et al. 2003)
Expanded clay	1.0-1.6	SiO_2, Al_2O_3, Fe_2O_3	398 (10-100 μm size)	(Occelli et al. 2002)
Pumice	0.4	Highly vesicular texture glass	2.1-14.2 (<63-1000 μm size)	(Kitis et al. 2007)

Table 1. Characteristics of different media used for water filtration

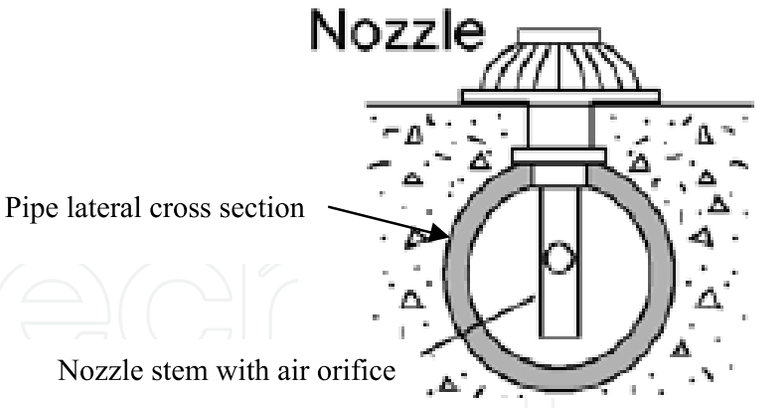

Figure 1. Sketch of typical nozzle design on a filter floor

To fully clean the filter media without forming dead zones, mud balls, media encrustation, and boiled media spots, even distribution of air and water during backwashing is critical. Fundamentally, there are two types of designs for simultaneous air and water distribution, nozzles and underdrain (with integrated dual channels). Nozzles (Figure 1) usually have slot sizes in the range 0.25–0.5 mm to minimize the risk of sand penetration. Nozzle arrangement density on the floor depends on the type of nozzles and is greater than about 35 nozzles/m^2 (Ratnayaka et al. 2009). However, the design of nozzle slots needs to be considered carefully to prevent fouling. There are several other underdrain systems, mostly of proprietary designs, successfully used in many parts of the world. An example is the design by Leopold (a Xylem brand) which comprises underdrain blocks (each block approximately 1 foot ×1 foot ×4 feet), formed from high-density polyethylene (HDPE), which snap lock together to form water resistant long laterals (Figure 2). The blocks incorporate a dual lateral design with a water recovery channel that ensures uniform distribution of concurrent air and water even over laterals up to 42 feet or 12.8 m. The blocks can be fitted with a porous HDPE IMS® Cap on top that helps to eliminate the need for support gravel (Figure 2).

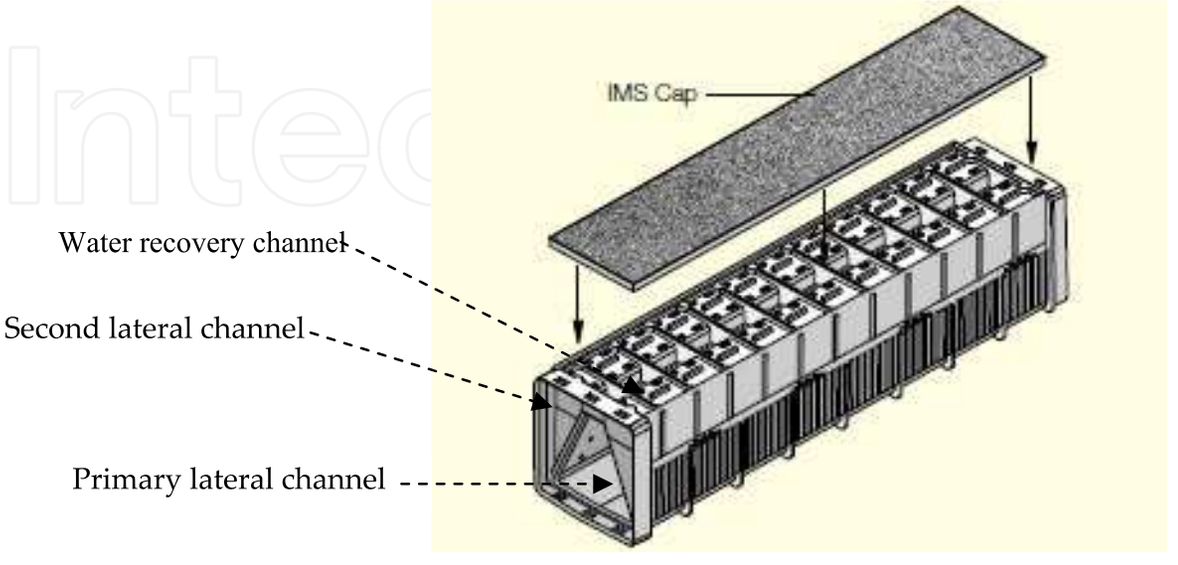

Figure 2. Sketch of typical underdrain design with dual channels (courtesy of Leopold, a Xylem brand)

2. Filtration with biological activity

Biological water and wastewater treatment processes are based on the growth of microbial communities capable of metabolizing contaminants through mediating oxidation-reduction reactions. The oxidants (electron acceptors) are normally oxygen, nitrate, perchlorate, sulfate, and Fe (III); the reductants (electron donors) are normally organic matter, trace organic compounds, ammonia, As (III), and iron (II) and Mn (II), etc. In a fixed-film biological process, biofilms are developed on media such as sand, anthracite, granular activated carbon (GAC), or membranes. A biofilm process mainly consists of two simultaneous steps, substrate diffusion and biological reaction, as illustrated in Figure 3. Electron donors and acceptors diffuse from bulk fluid into the biofilm and are metabolized by microbial cells in the biofilm, as a result of which the diffusion profiles are parabolic.

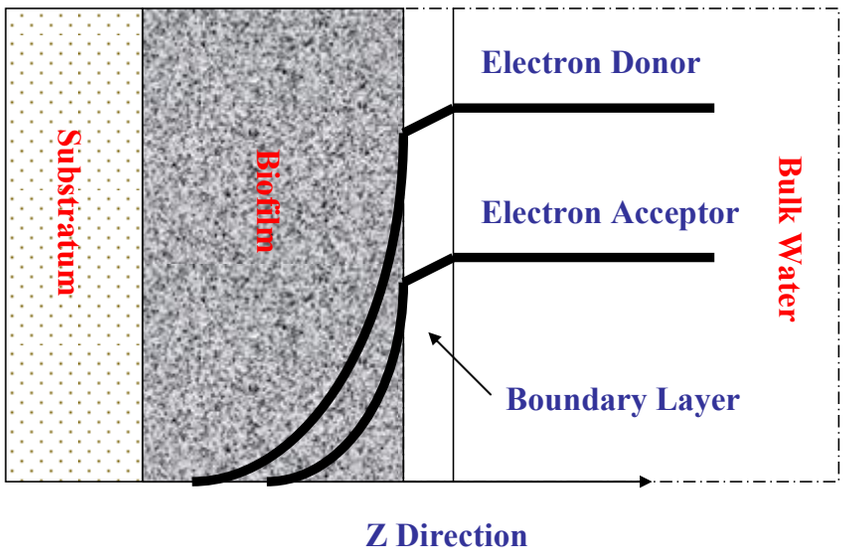

Figure 3. Schematic diagram of substrate diffusion in a biofilm attached to a solid substratum

Biologically active filters (BAFs) are essentially of the same physical structures as rapid gravity filters except BAFs are maximized with biological activities without backwashing the filters with chlorinated water or with no pre-chlorination. BAFs have been used for decades in North America and Europe in drinking water treatment, but have drawn more attention only recently. Regulatory and customer acceptance remain an issue because of the concern of microbial sloughing and breakthrough. A recent survey conducted by the AWWA Research Foundation indicated that 44% of the respondents believed biological processes in the drinking water industry were not accepted and 25% believed they were. Major operational concerns were breakthrough of pathogens and sloughing of bacteria (Evans et al. 2008). However, coliform bacteria were rarely observed in BAF effluents in laboratory studies, indicating that coliform organisms were eliminated by the microbial activity in the filters because of the competition for limited nutrients (Camper et al. 1985; Rollingger and Dott 1987). A pilot study demonstrated that biologically active filters reduced microbial activities in distribution systems (Characklis 1988). Furthermore, French experience indicated that removal of biodegradable materials resulted in a lesser amount of

microbial presence in the distributions systems (Bourbigot et al. 1982). Comparison of physicochemical and biological treatments indicated that biological treatment limited mutagenic generation (Carraro et al. 2000).

3. Applications and design parameters

Perhaps, slow sand filters (SSFs) were the earliest application of a biological process in drinking water treatment. The major function of a filter occurs at the surface layer (Schmutzdecke) of the sand bed which contains a zoogeal jelly in which biological activities are highest (Babbitt and Doland 1939). Full scale experience indicated NOM removal was 15±5 mg/L by slow sand filters (Collins et al. 1992). Coliform reduction was 2-4 log (Barrett et al. 1991), and E Coli reduction was found strongly correlated with carbon dioxide respiration in the top 2.5 cm media and protistan abundance in the top 0.5 cm Schmutzdecke (Unger and Collins 2008). With over 150 years of history, river bank filtration (RBF) showed efficient organic substance removal in full scale plants in the Netherlands and Germany (Piet and Zoeteman 1980; Sontheimer 1980). River bank filtration removed TOC by 33-86% and disinfection by-product formation potential by 30-100% at the wells in several drinking water utilities in the US (Partinoudi and Collions 2007; Weiss et al. 2003).

A previous review of biological processes in drinking water treatment summarized that a wide range of contaminants can be removed through biological oxidation and reduction of dissolved constituents including natural organic matter (NOM), ammonia, nitrate, perchlorate, and iron and manganese, where operating parameters were discussed (Bouwer and Crowe 1988). Additionally, BAFs were reported to remove trace organic compounds, halogenated organics, perchlorate, and arsenic. A BAF usually does not require the addition of other chemicals for oxidizing and removing of contaminants. It does not require close monitoring of a breakthrough point, as in conventional column adsorption processes. Some organics adsorbed in activated carbon particles can be degraded by microorganisms attached on the activated carbon, or through enzymatic reaction during normal operation and hence create some active adsorption sites (Perrotti et al. 1974; Rice and Robson 1982; Rodman et al. 1978). This process is referred as biological regeneration. The service life of activated carbon can be extended by biologically regenerating exhausted carbon. The treated water from the BAF is unlikely to produce undesirable disinfection by-products and bacteria re-growth in the water distribution system (Dussert and Van Stone 1994; Scholz and Martin 1997).

3.1. Removal of natural organic matter (NOM)

Natural organic matter (NOM), consisting of humic acid, fulvic acid, carbohydrates, and other natural compounds, is present in natural water sources and is a precursor of disinfection by products (DBPs). DBPs are compounds formed when strong oxidants such as chlorine and ozone come into contact with NOM. Epidemiological evidences supported an association between chlorinated water or trihalomethanes and bladder cancer (Cantor et al. 1987; Cantor et al. 1999; Doyle et al. 1997). The most prevalent DPBs that form as a result of contact between organic carbon and chlorine include total trihalomethanes (TTHMs) and

haloacetic acids (HAA5). To mitigate public exposure to these compounds, US Environment Protection Agency (EPA) has developed regulations restricting their concentrations at all points in the distribution system. The initial legislation formed for these restrictions is known as Stage 1 Disinfectants and Disinfection Byproducts Rule. A secondary stage for the DBP rule had also been promulgated by EPA in 2006. Stage 2 Disinfectants and Disinfections Byproducts Rule was applied as an addition to the continual improvement of safety in drinking water standards in the United States. Amendments to Stage 1 DBP rule include: (a) requiring annual averages at every point in the distribution system to adhere to the predefined maximum contamination levels (MCLs), (b) escalating the sampling frequency for communities with larger populations, and (c) the utility's distribution system must be evaluated to identify locations with elevated DBP concentrations.

To remove DBP precursors, biologically active filtration using different media was employed worldwide, usually assisted by pre-ozonation which increased assimilable organic carbon (AOC) or biodegradable organic matter (BOM) which was subsequently metabolized by biofilms in biofilters (Carlson and Amy 2001; Goel et al. 1995; Weiss et al. 2003). The design parameters are empty bed contact time (EBCT), media selection, media configuration, backwash regime selection, temperature in addition to pre-ozone doses. A summary of design parameters and media selection and observations in previous studies were provided in Table 2. Majority of the studies showed that an EBCT of 10 minutes should be used in process design to achieve 30-50% TOC removal with GAC. When combined with pre-ozonation, an EBCT of 5 minutes appeared sufficient (Hozalski et al. 1995; Laurent et al. 1999; Rittmann et al. 2002; Weiss et al. 2003).

The design of biologically active filters should be based on the achievement of one of the following criteria:

- Maximal removal of DOC to reduce the formation of DBPs;
- Maximal removal of BOM to minimize the risk of biological re-growth in the distribution system;
- Maximal removal of potential carcinogenic ozonated by-products (OBPs) such as formaldehyde.

Direct comparison showed that the GAC/sand filter produced better performance than the anthracite/sand filter (Rittmann et al. 2002) and the GAC/Sand filter out performed anthracite/sand filter by 11% for AOC removal (Weiss et al. 2003). GAC media showed better resistance to temperature at 1-3 ºC in terms of oxalate and TOC removal compared with anthracite (Emelko et al. 2006). The BOM (10% acetate and 90% other organic matter with a maximal degradation rate less than one-tenth that of acetate) removal was reduced from 55% at 22.5 ºC to 12% at 6 ºC in a sand filter at EBCT 7.5 minutes (Hozalski et al. 1999).

Emelko et al. (2006) studied the effect of backwash and temperature on full scale biofiltration, and concluded that biodegradable organic material (BOM) removal was not influenced by backwash regimes even though some biomass expressed by phospholipid was lost during backwash with air scouring. Others also concluded that backwashing did not

have noticeable impact on BOM removal because no more than 25% of biomass was washed out (Hozalski et al. 1999; Rittmann et al. 2002). However, microbial communities in the filters and during the operating condition shifts were not investigated in this study. In another study, it was found that backwashing caused changes in the relative compositions of microorganisms in a GAC biofilm in the top layer of the bed and reduced the attached bacterial abundance to 64% (Kasuga et al. 2007). The relative abundances of some terminal-restriction fragments (T-RFs) increased such as the *Planctomycetes*-derived fragment; however, some decreased, which included the *β-proteobacteria*-derived fragments (Kasuga et al. 2007).

Nutrient levels were also shown to influence the process efficiency. In a full scale study at Daugava water treatment plant in Riga, Latvia, the process including ozonation and biofiltration was not efficient for removal of dissolved organic carbon (DOC) from waters with a high amount of humic substances likely due to phosphorus limitation (Juhna and Rubulis 2004).

Phosphorus supplementation in a pilot study decreased biofilter terminal headloss by ~15 percent relative to the control likely the result of reduced EPS formation in the filter, and decreased contaminant breakthrough relative to the control biofilter, including MIB (~75 percent less breakthrough), manganese (~90 percent less breakthrough), and DOC (~15 percent less breakthrough) (Lauderdale et al. 2011).

Study Scales	Parameters	Findings	References
Pilot using filters of granular activated carbon (GAC), GAC/sand, anthracite/sand	GAC/sand filter: EBCT 5 minutes with 29% TOC removal; EBCT 10 minutes with 33% TOC removal; EBCT 15 minutes with 42% TOC removal; EBCT 20 minutes with 51% TOC removal 0.3-1.0 mg O_3/mg TOC	Pre-ozonation increased Assimilable Organic Carbon (AOC) in influent, but also increased BAF effluent AOC relative to non-ozonated influent water. GAC/Sand filter was better than anthracite/sand filter by 11% for AOC removal.	(Weiss et al. 2003)
IRWD pilot facility in Santa Ana, California, including BAF following ozonation.	EBCT from 3.5 to 9 minutes 1.0-1.8 g O_3/g TOC	Up to 90% of color was removed and up to 38% DOC was removed; GAC biofilter gave better performance than anthracite.	(Rittmann et al. 2002)
Laboratory-scale batch degradation tests	0-7.3 mg O_3/mg TOC	Biodegradability of four NOM sources was improved by ozonation in the range of 0-7.3 mg	(Goel et al. 1995)

Study Scales	Parameters	Findings	References
		O_3/mg TOC. Degradation of high molecular weight organics were more influenced by ozonation	
Laboratory-scale biologically active sand (ES 0.5 mm) filters.	EBCT from 4 to 20 minutes 2-4 mg O_3/mg TOC	Ozonated NOM removal was significantly affected by the sources of the organic carbon independent of EBCT	(Hozalski et al. 1995)
Laboratory-scale biodegradability study with Ohio River	0.6-1.0 mg O_3/mg TOC (optimal for DBP reduction) 2.0 mg O_3/mg TOC (maximal for AOC)		(Shukairy et al. 1992)
Laboratory-scale biologically active glass beads and sand (ES 0.52 mm) filters.	EBCT 7.5 minutes 0.58±0.12 mg O_3/mg TOC	30% TOC removal was achieved. Perfromance was not impaired by backwash.	(Hozalski et al. 1999)
Pilot including biologically active filters (expanded clay 0.5-2.5 mm) following ozonation.	EBCT 11-54 minutes 1.0-1.7 mg O_3/mg TOC	EBCT did not have a significant impact; TOC removal 18-37% ; majority (80%) of BOM were removed;	(Melin and Odegaard 1999)
Full scale GAC filters at River dune Water Works	EBCT 20 minutes with two stages 0.35-0.45 mg O_3/mg TOC	50% TOC removal; successive reactivation of GAC was still effective	(van der Hoek et al. 1999)
Full scale BAC filters, St-Rose Treatment plant, Canada	EBCT 5-12 minutes	50% BDOC removal	(Laurent et al. 1999)

Study Scales	Parameters	Findings	References
Lab scale biodegradability study	2 mg O_3/mg TOC	40-50% DOC removal 40-60% THMFP removal 90-100% aldehydes removal	(Siddiqui et al. 1997)
Pilot including biologically active anthracite filters following ozonation.	0.6 mg O_3/mg DOC EBCT 2-11 minutes	Maximal 9% DOC was removed at EBCT 15 minutes with 5 m/h and EBCT 7 minutes with 9.7 m/h; 80% ozone by-products were removed at EBCT 3-5 minutes	
Lab scale fluidized GAC filter	EBCT 20 minutes 1 mg O_3/mg TOC	45% BDOC removal	(Yavich and Masten 2003)
Pilot including biologically active anthracite filters following ozonation.	0.6 mg O_3/mg DOC EBCT 6 minutes Temperature 6-10 ºC	Up to 1.0 mg O_3/mg DOC produced maximal $BDOC_{rapid}$/DOC; and 0.4-0.6 mg O_3/mg DOC produced maximal $BDOC_{rapid}/BDOC_{total}$. Cumulative 90% DOC removal at EBCT 6 minutes.	(Carlson and Amy 2001)
Pilot including anthracite and GAC filters directly following ozonation.	EBCT 5 minutes 0.5-1.0 mg O_3/mg DOC	7-9% TOC (as UV254) removal; No difference was observed between GAC and anthracite filters due to the nature of the water	(Chaiket et al. 2002)
Full scale including GAC filters directly following ozonation at Sweeney WTP, Wilmington, NC.	EBCT 10-60 minutes 0.5-2 mg O_3/mg DOC	10-50% DOC removal; no significant difference between lignite and bituminous GAC.	(Najm et al. 2005)

Table 2. Design parameters and findings for the removal of natural organic matter removal (NOM) in previous studies

3.2. Removal of MIB and geosmin

The presence of tastes and odors in drinking water is an increasing and serious problem in the United States and the world. Some species of algae and bacteria naturally produce odorous chemicals inside their cells. Geosmin (trans-1, 10-dimethyl-trans-9-decalol) and MIB (2-methylisoborneol) are common odorous chemicals. There are no maximal contamination levels (MCLs) for MIB and Geosmin in drinking water systems according to US EPA. However, earthy and musty odors generated by Geosmin and MIB are detectable by individuals at the concentrations of 5 to 10 parts per trillion, and often result in customer complain. When large numbers of algae and bacteria flourish in a water body (an "algae bloom"), taste and odor-compound concentrations increase to levels above this threshold and cause taste and odor problems.

Biological active filtration was effectively used for the removal of Geosmin and MIB as summarized in Table 3. Since the concentrations of Geosmin and MIB encountered in drinking water systems are usually much less than that of TOC, a secondary utilization pathway was proposed as opposed to primary substrate utilization (Bouwer and Crowe 1988). Primary substrates support steady state biofilms which in turn metabolize secondary substrates such as Geosmin and MIB.

Unlike the removal of TOC, placing ozonation in front of GAC filters was not benefiting the removal of MIB and Geosmin likely due to the competition from increased AOC (Vik et al. 1988). An important design parameter is EBCT, which is usually in the range of 5-20 minutes depending on the required removal. GAC filters provided resistance to temperature variation while the removal was reduced by 24% for both Geosmin and MIB when temperature was reduced from 20 to 8°C in anthracite filters (Elhadi et al. 2006). At lower temperatures (6-12 °C), MIB and Geosmin removal was also reduced with expanded clay by 15% and 10%, respectively, compared to that at 15 °C (Persson et al. 2007). Biodegradation of both MIB and Geosmin was determined to be a pseudo-first-order reaction, with rates influenced by the initial amount of the biofilm biomass (Ho et al. 2007). As a result, sand with a well-established biofilm taken from a 26 years old filter was capable of removing MIB and Geosmin to below detection limits after 11 days of operation while sand without an established biofilm removed 60% Geosmin and 40% MIB after 154 days of operation (McDowall et al. 2007). Four bacteria, a *Pseudomonas sp.*, an *Alphaproteobacterium*, a *Sphingomonas sp*. and an *Acidobacteriaceae* member were identified as microorganisms most likely involved in the biodegradation of Geosmin within the sand filters (Ho et al. 2007).

Study Scales	Parameters	Findings	References
Full scale study at CLCJAWA Water Treatment Plant at Lake Bluff, Illinois, which included biologically active GAC filters following ozonation.	0.66-0.81 mg O_3/mg TOC EBCT 10-20 minutes	Ozonation removed 36-65% MIB and biofiltration removed 26-46% of MIB. The biodegradability of geosmin and MIB was confirmed by a bench scale study, where 55% and 44% removal was achieved for geosmin and MIB, respectively.	(Nerenberg et al. 2000)

Study Scales	Parameters	Findings	References
Pilot study of sand filters capped with biologically active GAC. Geosmin was in the range of about 70 to 110 ng/L,	EBCT 5 minutes	11 to 38%	(Ndiongue et al. 2006)
	EBCT 7.5 minutes	78% geosmin removal; Geosmin was better removed than MIB.	
Bench-scale two 2.0 m high glass GAC/sand filters	EBCT 5.6 minutes Temperature 12-16 °C	76 to 100% geosmin removal and 47% to 100% MIB removal. The exhausted GAC initially removed less geosmin and MIB, but the removals increased over time.	(Elhadi et al. 2004)
Bench-scale GAC/sand and anthracite/sand filters	EBCT 5.6 minutes Temperature 8 and 20 °C	60% geosmin and 40% MIB removal at 20 °C in GAC filters. 36% geosmin and 16% MIB removal at 8 °C in anthracite filters.	(Elhadi et al. 2006)
Bench-scale sand filters	EBCT 15 minutes	60% geosmin and 40% MIB with new sand after 154 days; reduced to below detection limit with sand from a 26 years old filter with a well-established biofilm	(McDowall et al. 2007)
Pilot study of GAC and expanded clay filters	EBCT 6, 15 and 30 minutes	Exhausted GAC had adsorption capability for MIB and Geosmin. At initial 20 ng/L MIB and 20 ng/L Geosmin: 97% removal at 30 minutes EBCT; 90% removal at 15 minutes EBCT; >40% remocal at 6 minutes EBCT	(Persson et al. 2007)
Bench-scale sand filters	EBCT 15 minutes 20±2 °C	95% removal of both MIB and geosmin with sand from an over 30 year facility	(Ho et al. 2007)
Pilot GAC and ozone plus GAC	EBCT 21 minutes 2-5 O_3 mg/L 1.5-4 TOC mg/L	TOC removal was better with ozone plus GAC; GAC was better for MIB and geosmin removal than ozone plus GAC because of the competition of TOC. GAC kept Geosmin and MIB below 10 ng/L	(Vik et al. 1988)

Table 3. Design parameters and findings for the removal of MIB and Geosmin in previous studies

3.3. Removal of iron, manganese, and arsenic

Appreciable amounts of iron and manganese usually exist in ground water or lake water experiencing low dissolved oxygen levels. The US EPA set secondary MCLs for iron and manganese at 0.3 mg/L and 0.05 mg/L, respectively.

There are two valences of iron and manganese, Fe (II) and Fe (III); Mn (II) and Mn (IV). Fe (II) and Mn (II) are quite soluble than Fe (III) and Mn (IV), respectively. As summarized in a previous study, the solubility product of ferrous hydroxide was in the range of 7×10^{-13} to 4.5×10^{-21} while the solubility product of ferric hydroxide was 3×10^{-38} to 4×10^{-36} (Gayer and Woontner 1956). The solubility product of manganous hydroxide was 9.0×10^{-14} and manganese dioxide was in equilibrium with $Mn(OH)_4$ (aq) with an equilibrium constant of 4.0×10^{-5} (Swain et al. 1975). $Mn(OH)_4$ (aq) will be prone to adsorption during filtration. Therefore, media filtration will not be effective to remove total iron and manganese if considerable portions are at the lower valance. Physicochemical removal requires a strong oxidant injected in front of media filtration to oxidize lower valance metals to a higher valance (Equations 1 and 2) and then filtered out.

$$2Fe^{2+} + 3/2O_2 = Fe_2O_3 \quad (1)$$

$$Mn^{2+} + O_2 = MnO_2 \quad (2)$$

Fe (II) and Mn (II) can provide energy as electron donors for autotrophic biological reactions when oxygen is present. Biological filtration was demonstrated effective for iron and manganese removal assisted with aeration or ozonation in front of filtration (Table 4). It appeared that EBCT of 10 minutes reduced iron and manganese by 95-100% with coarse or fine sand media (Katsoyiannis and Zouboulis 2004; Lytle et al. 2007a; Štembal et al. 2005; Tekerlekopoulou et al. 2008). The use of ozone was beneficial (Pokhrel et al. 2005), but it appeared that aeration was sufficient for providing oxygen (Katsoyiannis and Zouboulis 2004; Lytle et al. 2007a; Štembal et al. 2005; Tekerlekopoulou et al. 2008).

Arsenic is a semi-metal element in the periodic table. It is odorless and tasteless. It enters drinking water supplies from natural deposits in the earth or from agricultural and industrial practices. Non-cancer effects can include thickening and discoloration of the skin, stomach pain, nausea, vomiting; diarrhea; numbness in hands and feet; partial paralysis; and blindness. Arsenic has been linked to cancer of the bladder, lungs, skin, kidney, nasal passages, liver, and prostate. US EPA set the arsenic standard for drinking water at 0.01 mg/L (10 parts per billion or ppb) to protect consumers served by public water systems subject to the effects of long-term, chronic exposure to arsenic.

Understanding the oxidation state is important for arsenic removal from drinking water. There are two oxidation states for arsenic: arsenite (As (III)) and arsenate (As (V)). Arsenite typically forms aqueous $As(OH)_3$, $As(OH)_4^-$, and AsO_2OH^{2-}, depending on pH; dissolved arsenate forms AsO_4^{3-} , $HAsO_4^{2-}$, or $H_2AsO_4^-$ (Edwards 1994; Katsoyiannis et al. 2002). At $6.9 < pH < 11.5$, $HAsO_4^{2-}$ is the primary species; and at $2.2 < pH < 6.9$, $H_2AsO_4^-$ is the primary arsenate species (Edwards 1994; Katsoyiannis et al. 2002). Arsenate adsorbs to soil minerals, particularly iron oxides and hydroxides. Arsenate sorption to iron oxides peaks around pH

5-7. Arsenite tends to adsorb less strongly than arsenate. Source water containing arsenite generally requires using a strong oxidant, e.g., chlorine, chlorine dioxide, and ozone, to oxidize arsenite to arsenate which can be removed by coagulation and filtration. Arsenic (V) removal by either ferric chloride or alum was relatively insensitive to variations in source water composition below pH 8 meanwhile arsenic (III) removal by ferric chloride was less efficient and more strongly influenced by source water composition than arsenic(V) removal (Hering et al. 1997). The presence of sulfate (at pH 4 and 5) and natural organic matter (at pH 4 through 9) adversely affected the efficiency of arsenic (III) removal by ferric chloride and arsenic(III) could not even be removed by coagulation with alum (Hering et al. 1997).

Study Scales	Water Qualities	Parameters	Findings	References
Pilot GAC column with 1740 mm (height) by 450 mm (diameter)	Fe 6.4-8.4 mg/L Mn 0.93-0.99 mg/L As 14.5-27.3 μg/L DOC 4.3-4.9 mg/L	EBCT 34.1 minutes 6.2-8.5 O_3 mg/L	The biologically active system removed 99.8% of Fe(II), and with the ozone pretreatment, the average removal increased to 99.9% for Fe(II).	(Pokhrel et al. 2005)
Pilot study of roughing filter with gravel and sand filter	Fe 0.09-0.44 mg/L Mn 0.18-1.83 mg/L	EBCT 30 and 60 minutes (roughing filter) EBCT 5 minutes (sand filter)	Iron and manganese removal efficiencies were between 85% and 95%.	(Pacini et al. 2005)
Pilot filter (1 m high polystyrene Beads 3-4 mm)	Fe 2.8 mg/L	EBCT 7.3 minutes	Fe(II) was microbiologically oxidized to Fe(III) precipitated on the filter bed.	(Katsoyiannis and Zouboulis 2004)
Bench-scale sand filter	Mn 0.86-1.83 mg/L		90% Mn removal	(Burger et al. 2008)
Pilot trickling filter (1.9 mm sand)	Mn 0.6–2.0 mg/L	EBCT 9 minutes	Close to 100% Mn removal	(Tekerlekopoulou et al. 2008)
Pilot sand filter (190 cm high, 0.5-2.0 mm sand)	Fe 0-2.45 mg/L Mn 0.1-1.06 mg/L NH_4-N 0.02-2.62 mg/L	EBCT 4.75-10.4 minutes	95% Fe, Mn, and NH_4-N removal	(Štembal et al. 2005)
Pilot dual media filter (20″ anthracite over 10″ sand)	As 37±2 μg/L NH_4-N 1.15 mg/L Fe 2289 ±114 μg/L	EBCT 9.4 minutes	Reduced Fe to less than 25 μg/L	(Lytle et al. 2007a)

Table 4. Design parameters and findings for the removal of iron and manganese in previous studies

Study scale dual media filters at	Water Qualities	Parameters	Findings	References
Pilot dual media filter (20″ anthracite over 10″ sand)	As 37±2 μg/L NH_4-N 1.15 mg/L Fe 2289 ±114 μg/L	EBCT 9.4 minutes	Reduced to less than 10 μg/L; Majority of As in the effluent was particular.	(Lytle et al. 2007a)
Pilot anthracite/sand filters	As 20-46 μg/L	EBCT 10 minutes	Reduced to less than 10 μg/L	(Lytle et al. 2007b)
Pilot GAC column with 1740 mm (Height) by 450 mm (Diameter).	Fe 6.4-8.4 mg/L Mn 0.93-0.99 mg/L As 14.5-27.3 μg/L DOC 4.3-4.9 mg/L	EBCT 34.1 minutes 6.2-8.5 O_3 mg/L	97% without ozonation; 99% with pre-ozonation	(Pokhrel et al. 2005)
Pilot filter (1 m high polystyrene Beads 3-4 mm)	As 50–200 μg/L	EBCT 7.3 minutes	Reduced to less than 10 μg/L	(Katsoyiannis and Zouboulis 2004)
Pilot filter (1 m high polystyrene Beads 3-4 mm)	As 40–50 μg/L DO 2.7 mg/L	EBCT 7.3 minutes	80% As removal	(Katsoyiannis et al. 2002)
Pilot filter with at least 0. 66 m sand	As(III) 30-200 μg/L Fe(II) 0.5-1.5 mg/L Mn(II) 0.6-2.0 mg/L	EBCT 7.9 minutes	95% As removal	(Liu et al. 2010)
Pilot slow sand filter (sand 0.45-0.55 mm mixed with iron fillings)	As 50 μg/L	EBCT 4-5 hours 0.023 m/h	Columns containing filtration sand only removed As <11%; all iron/sand columns achieved greater than 92% removal.	(Gottinger 2010)
Pilot slow sand filter (90 cm, sand 0.25-2 mm)	As 10–35 μg/L	EBCT 4.25 hours	Reduced to less than 5 μg/L if Fe/As feed ratio was 40	(Pokhrel and Viraraghavan 2009)
Pilot GAC filter (100 cm, GAC 2-4 mm)	As(III) 25 mg/L Fe(II) 10 mg/L Mn(II) 2 mg/L	EBCT 6 hours	>99% As removal 80% Fe removal 95% Mn removal	(Mondal et al. 2008)

Table 5. Design parameters and findings for the removal of arsenic in previous studies

The current treatment options include activated alumina, iron oxide coated sand, greensand, reverse osmosis, ion exchange, and electrodialysis in addition to coagulation (Edwards 1994).

Similar to Fe (II) and Mn (II), arsenite can provide energy as an electron donor for autotrophic biological reactions when oxygen is present. Biological filtration was shown effective for arsenite removal assisted with aeration or ozonation in front of filtration (Table 5). Various media including sand, anthracite, GAC, and polystyrene beads were used for arsenic removal. Generally an EBCT of 10 minutes is required to reduce arsenic from up to 100 μg/L down to less than 10 μg/L. Due to the strong affinity of arsenate to ferric oxide, feeding ferric in the influent increased the removal efficiency with an increasing Fe/As ratio (Pokhrel and Viraraghavan 2009). A study found filtration columns containing mixture of sand and iron fillings improved removal and were capable of reducing arsenic from 50 to well below 10μg/L with an average of 92% removal (Gottinger 2010).

To improve the removal efficiency, immobilizing whole bacterial cells has attracted more research interest in recent years. *Ralstonia eutropha MTCC 2487* (this strain can produce ArsR protein and arsenate reductase enzyme) was immobilized on GAC bed in the column reactor (Mondal et al. 2008). *D. Desulfuricans*, *G. ferrigunea*, *L. ocracia*, *R. picketti*, *T. ynys1*, *Gallionella,* and *Leptothrix* were exploited to remove arsenic in biofiltration columns (Brunet et al. 2002; Elliot et al. 1998; Jong and Pany 2003; Katsoyiannis et al. 2002).

3.4. Removal of ammonia (nitrification)

Although there is no ammonia drinking water standard in the United States, the European community has established a maximum limit of approximately 0.5 mg/L and a guide level of 0.05 mg/L (EU Council 1980). Although there are no immediate indications that ammonia will become regulated within the United States, there are benefits for utilities to reduce the amount of ammonia that is able to enter a distribution system. The presence of ammonia in drinking water distribution systems has been correlated to increased biological activity, corrosion, formation of nitrite and nitrates, and adverse impacts on taste and odor (AWWA 2006). In addition, the presence of ammonia can interfere with the effectiveness of some water treatment processes including biological manganese removal as ammonia removal must be achieved before manganese removal due to the fact that the oxidation potential for nitrification is lower than manganese oxidation (McGovern and Nagy 2010).

Nitrification can be achieved in different ways, but may be most cost effectively accomplished by employing biofiltration (Table 6). The effectiveness of biological ammonia oxidation treatment to reduce source water ammonia levels is dependent on a number of source water and engineering design factors including temperature, dissolved oxygen, TOC, pH, biomass quantity and population, media type, and surface area, as well as hydraulic loading rate and contact time (Zhang et al. 2009). Factors affecting nitrification occurrence, nitrification impacts on water quality and corrosion, and nitrification monitoring and control methods were reviewed previously (Zhang et al. 2009). Arrhenius coefficient was

1.12 without acclimation and 1.06 with acclimation (Andersson et al. 2001). At temperature less than 4 °C, nitrification seemed un-sustained and feeding low temperature culture (psychrophiles) seemed necessary (Andersson et al. 2001). However, in fixed-film biofilters, the impact of temperature on nitrification rate was less significant than that predicted by the van't Hoff-Arrhenius equation, and a temperature increment at 20 ºC resulted in a nitrification rate increase of 1.108% per degree and 4.275% per degree under DO and ammonia limiting conditions, respectively (Zhu and Chen 2002).

Study Scales	Water Qualities	Parameters	Findings	References
Pilot sand filter (190 cm high, 0.5-2.0 mm sand)	Fe 0-2.45 mg/L Mn 0.1-1.06 mg/L NH_4-N 0.02-2.62 mg/L	EBCT 4.75-10.4 minutes	95% Fe, Mn, and NH_4-N removal	(Štembal et al. 2005)
Full scale and pilot scale GAC filters at St Rose WTP at Laval, Canada	NH_4-N < 1 mg/L	EBCT up to 28 minutes	EBCT 5 minutes seems suitable for removing ammonia;	(Andersson et al. 2001)
Pilot gravel filter (180 cm high, 6-12 mm and 12-25 mm gravel)	NH_4-N < 1 mg/L 26 °C	EBCT 4-23 minutes	70-80% at EBCT 7.8 minutes; >90% at EBCT 10 minutes	(Forster 1974)
Pilot aerated GAC filter (85 cm high)	NH_4-N 2.88 mg/L	EBCT 10 minutes	95% ammonia removal	(Rogalla et al. 1990)
Pilot dual media filter (20" anthracite over 10" sand)	NH_4-N <1.7 mg/L	EBCT 9.4 minutes	Close to 100% ammonia removal	(Lytle et al. 2007a)

Table 6. Design parameters and findings for the removal of ammonia in previous studies

European experience on nitrification was reviewed for trickling filters, up-flow fluidized bed filters, rapid sand filters, and GAC filters (Rittmann and Snoeyink 1984). In a pilot trickling filter operated at 2.4 m/h with 2 m gravel media, ammonia removal was 80% at 20 ºC; 78% at 15 ºC; 67% at 10 ºC; and 50% at 5 ºC. In fluidized filters, nearly 100% removal was achieved as long as the fluidized solids were at least 30% by volume from 4-20 ºC. Nearly 100% nitrification was achieved using rapid sand filters at Mulheim where raw water contained 1 mg/L ammonia nitrogen. Complete nitrification was achieved with GAC filters at 10 m/h with an EBCT of 10 minutes.

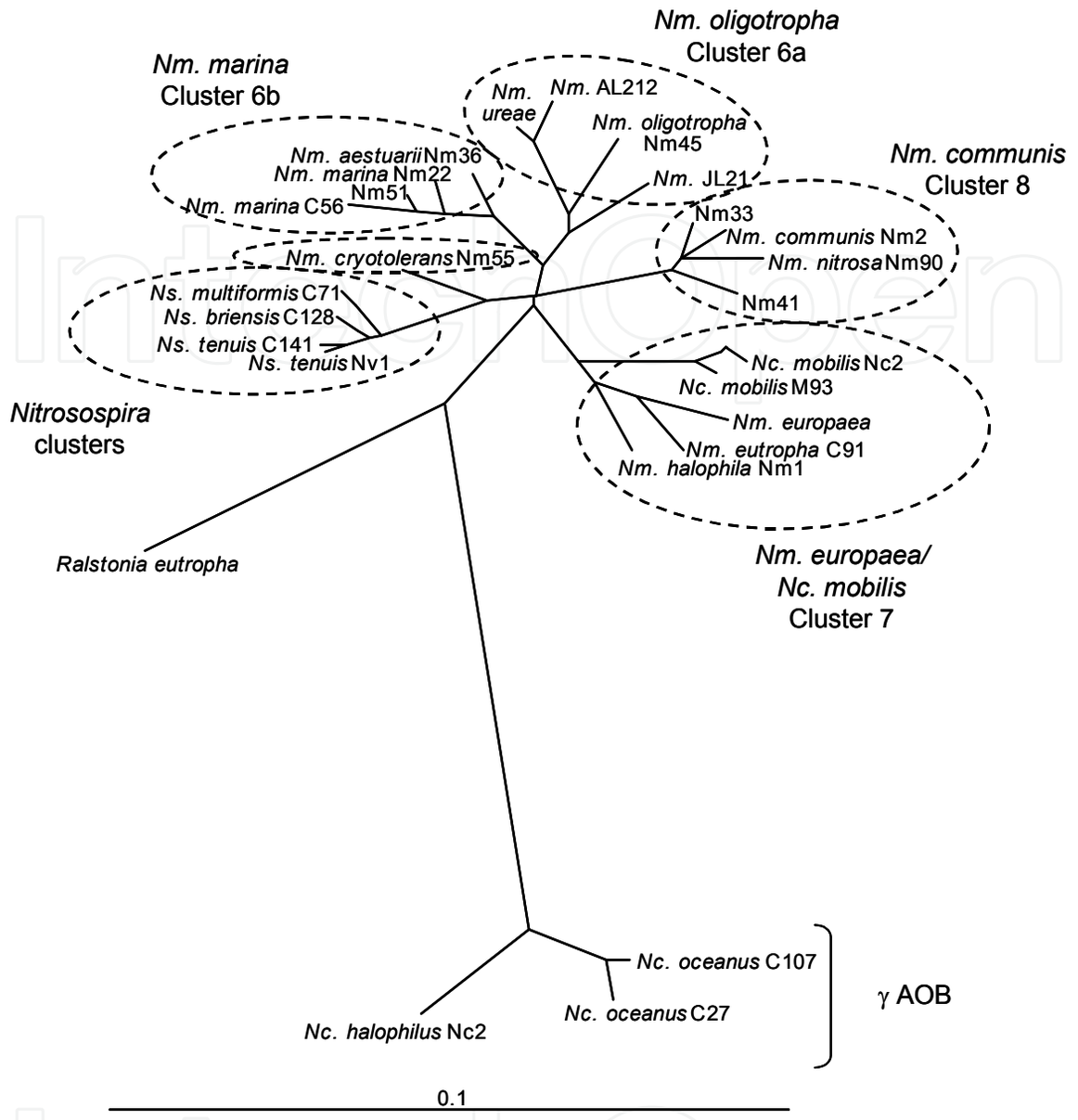

Figure 4. Phylogenetic tree of AOB based on multiple alignment of 55 nearly full-length AOB 16S rDNA sequences. Abbreviations are Nm for *Nitrosomonas*, Nc for *Nitrosococcus*, and Ns for *Nitrosospira*. *R. eutropha* is a non-AOB member of the *Betaproteobacteria* subphylum. Scale bar represents 10% sequence difference (Regan 2001)

Nitrification is a two-step process: ammonia oxidation and nitrite oxidation. The bacterial genera associated with ammonia oxidation are named as ammonia oxidizing bacteria (AOB) and the bacterial genera associated with nitrite oxidation are named as nitrite oxidizing bacteria (NOB). Both AOB and NOB are autotrophic bacteria using carbon dioxide for cellular synthesis under aerobic conditions. Phylogenetic trees for AOB and NOB were summarized elsewhere (Regan 2001) as shown in Figures 4 and 5, respectively. Kinetic parameters including specific substrate utilization rate, half saturation constant, yield, etc of *Nitrosomonas* and *Nitrobacter* were summarized at different temperatures (Rittmann and Snoeyink 1984).

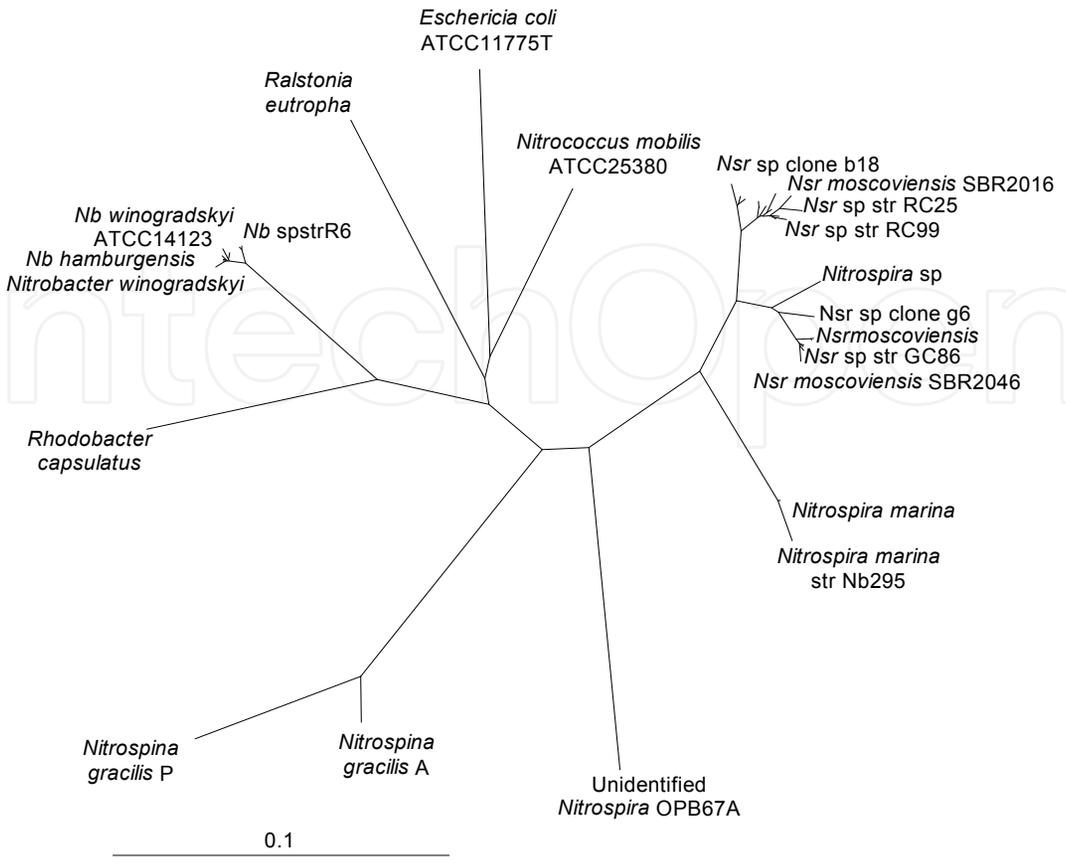

Figure 5. Phylogenetic tree of NOB based on a multiple alignment of 40 NOB 16S rDNA sequences. Abbreviations are Nb for *Nitrobacter* and Nsr for *Nitrospira*. *Rh. capsulatus* is in the *Alphaproteobacteria* class, *R. eutropha* is in the *Betaproteobacteria* class, and *E. coli* is in the *Gammaproteobacteria* class. Scale bar represents 10% sequence difference (Regan 2001)

3.5. Removal of nitrate (denitrification) and perchlorate

With the application of nitrate containing fertilizers and consumption of animal products, more nitrate is discharged into rivers and lakes, which may cause eutrophication and elevated levels of nitrate in ground water and surface water. Although nitrate was not identified as a carcinogen to laboratory animals, methaemoglobinaemia forms as a consequence of the reaction of nitrite (reduced from nitrate in human bodies) with haemoglobin in the human red blood cells to form methaemoglobin, which binds oxygen tightly and does not release it, therefore blocks oxygen transport (WHO 2008). The maximal contamination level in drinking water was 10 mg/L nitrate nitrogen in US, Japan, and Korea. The European Union countries set the standard for nitrate nitrogen at 11.3 mg/L. World Health Organization recommends 11.3 mg/L nitrate nitrogen to protect against methaemoglobinaemia in bottle-fed infants. Traditionally, nitrate removal was achieved by biological denitrification, ion-exchange, adsorption, chemical reduction, and membrane separation such as reverse osmosis. Ion exchange and membrane processes were often applied for high purity water treatment, which will generate concentrated nitrate reject (usually need additional treatment) from resin or membranes. Biological treatment processes were widely used for wastewater and drinking water treatment.

Denitrification filters are a unique type of biologically active filter where an external carbon source is usually added to the filter to provide a food source to anoxic biology and facilitate the reduction of nitrates in the filter. As a result, a dedicated anoxic biology is likely developed in the filter.

It is generally recognized that denitrification is carried out in the following steps with the aid of various enzymes produced during the process in the form of intracellular and extracellular polymeric substances, i.e., nitrate reductase (Nar), nitrite reductase (Nir), nitric oxide reductase (Nor), and nitrous oxide reductase (Nos) (Rittmann and McCarthy 2001).

$$NO_3^- - N \xrightarrow{Nar} NO_2^- - N \xrightarrow{Nir} NO \xrightarrow{Nor} N_2O \xrightarrow{Nos} N_2$$

Denitrifiers are a group of heterotrophic bacteria and are phylogenetically diverse. They belong to over 50 genera and fall into all major physiological groups (Zumft 1992; Zumft 1997). Furthermore, microorganisms fed with different carbon sources showed distinct features. A report demonstrated that the metabolic profiles obtained from potential denitrification rates with 10 electron donors were altered with their preferences for certain compounds after supplementing methanol or ethanol, and that methanol had the greater impact (Hallin et al. 2006).

Denitrifying bacteria fed with methanol were recognized as methylotrophs. Fluorescence in situ hybridization (FISH) combined with microautoradiography (MAR) revealed that *α-Proteobacteria* metabolized ^{14}C methanol in the presence of nitrate, suggesting their involvement in denitrification in a methanol-fed fluidized marine denitrification reactor (Labbe et al. 2007). Using a molecular tool, *Paracoccus* sp. and *Hyphomicrobium* spp. were identified as denitrifiers in a denitrification sand filter fed with methanol (Neef et al. 1996). Research showed that methanol denitrifiers and acetate denitrifiers were distinctively different. When acetate was used as the carbon source, 16S rRNA gene sequences obtained from ^{13}C-labeled DNA were closely related to the 16S rRNA genes of *Comamonadaceae* and *Rhodocyclaceae* of the β-*Proteobacteria*, and *Rhodobacteraceae* of the α-*Proteobacteria*. When methanol was used as the carbon source, 16S rRNA gene sequences retrieved from ^{13}C-DNA were affiliated with *Methylophilaceae* and *Hyphomicrobiaceae* (Osaka et al. 2006).

A study showed that methanol-utilizing organisms can not use acetate or sugar (at least not immediately). Adding alternative carbon sources, i.e., acetate or sugar, will not result in an immediate improvement in denitrification (Dold et al. 2008). However, the substrate uptake rate (μ_{max}) and specific denitrification rate (SDNR), measured by feeding ethanol to methanol-utilizers, indicated that ethanol was also used essentially as easily and at a similar rate to methanol by the methanol-utilizers (Dold et al. 2008).

The stoichiometric and kinetic information for different carbon sources commonly used in a denitrification filter were studied previously and selected parameters are summarized elsewhere (Omnis-Hayden and Gu 2008). Table 7 summarized findings and operating parameters in previous studies. The important design parameters are EBCT, C/N ratio (the ratio of external carbon to nitrate nitrogen), pH and temperature.

Study Scales	Water Qualities	Parameters	Effluent Qualities, Efficiencies, and Findings	References
Pilot up-flow fluidized sand filter (1.2 m deep 0.6-0.8 mm sand)	NO_3-N 27 mg/L	EBCT 2.57 minutes C/N 1.3 (propionic acid)	Close to 100% NO_3-N removal	(Holló and Czakó 1987)
Pilot Rotating Biological Contactors (RBC) (75 m^2)	NO_3-N 40-250 mg/L pH 7 and 20±2 ºC	NO_3-N loading: 76 mg/m^2•h 490 mg/m^2•h Acetic acid as a carbon source	99% (for 76 mg/m^2•h) and 87% (490 mg/m^2•h) The optimum C/N ratio 4.3	(Mohseni-Bandpi et al. 1999)
Bench-scale membrane reactors.	NO_3-N 100-200 mg/L 20±1 ºC	C/N ratio 2-4 (ethanol)	99% removal with denitrification rates up to 1.23 g NO_3-N/m^2 •d	(Fuchs et al. 1997)
Bench-scale up-flow fixed-bed reactor	NO_3-N 20 mg/L 25±1 ºC	EBCT 48 minutes Molasses	85% NO_3-N removal with an optimum C/N ratio 2	(Ueda et al. 2006)
Bench-scale experiments. Calcium tartrate (4% w/w) co-immobilized in alginate beads with microorganisms	NO_3-N 110 mg/L		99% NO_3-N removal with NO_2-N residual (2-4 mg/L) higher than expected	(Liu et al. 2003)
Bench-scale carbon packed fixed-biofilm bed reactor, inoculated with *Paracoccus denitrificans* NRRL B-3784.	NO_3-N 200 mg/L	EBCT 78 minutes C/N ratio 2-4 (ethanol)	90% NO_3-N removal	(Pekdemir et al. 1998)
Bench scale packed Pall Rings (16 mm diameter and length) bed	NO_3-N 50 mg/L pH 7.5-7.8	EBCT 30 minutes COD/N ratio 4 (ethanol)	NO_3-N 0.9 mg/L; NO_2-N 0.7 mg/L; COD 34 mg/L	(Dahab and Kalagiri 1996)
Bench scale packed Pall Rings (16 mm diameter and length) bed	NO_3-N 50 mg/L pH 7 and 20 ºC	EBCT 30 minutes COD/N ratio 4 (ethanol)	NO_3-N 2.4 mg/L; NO_2-N 0.8 mg/L; COD 18 mg/L	(Woodbury and Dahab 2001)
Full scale packed polypropylene beads (3.8 mm and 5 mm diameter) bed	NO_3-N 20 mg/L pH 7.2 and 13-18 ºC	EBCT 66 minutes COD/N ratio 5.3 (corn syrup)	NO_3-N 5.0 mg/L; NO_2-N 1.7 mg/L; COD 20 mg/L	(Silverstein and Carlson 1999)

Study Scales	Water Qualities	Parameters	Effluent Qualities, Efficiencies, and Findings	References
Pilot ceramic media bed (1.5 m high and 0.3 m diameter)	NO_3-N 68 mg/L pH 7-7.5 and 15-20 ºC	EBCT 72 minutes COD/N ratio 4.3 (ethanol)	NO_3-N <4 mg/L; NO_2-N<3 mg/L; COD 10 mg/L	(Moreno et al. 2005)
Bench scale PVC/GAC beads bed (88 cm high and 12 cm diameter)	NO_3-N 45 mg/L and 20 ºC	EBCT 306 minutes COD/N ratio 5.5 (acetate)	NO_3-N 5 mg/L; NO_2-N<0.5 mg/L; COD 60 mg/L	(Vrtovsek and Ros 2006)
Bench scale moving bed (Kaldnes K1)	NO_3-N 60 mg/L pH >7 and 20 ºC	EBCT 54 minutes COD/N ratio 13.1 (acetate)	NO_3-N 4.7 mg/L; NO_2-N<0.25 mg/L; COD 400 mg/L	(Welander and Mattiasson 2003)
Pilot 900 L moving bed reactor (carrier, Natrix 6/6C, ANOX AB, Lund)	NO_3-N 800 mg/L pH 7.8 and 17 ºC	EBCT 17 hours COD/N ratio 4 (acetate)	NO_3-N ~0 mg/L; NO_2-N ~0 mg/L	(Welander et al. 1998)
Pilot moving plastic media bed	NO_3-N 13 mg/L NO_2-N 0.5 mg/L 7-10 ºC	EBCT 26 minutes COD/N ratio 4 (acetate)	NO_3-N 2.0 mg/L; NO_2-N 0.9 mg/L; COD 50 mg/L	(Rusten et al. 1995)
Membrane bioreactor with hollow-fibers (1.1 mm inner diameter, 1.4 mm external diameter, 0.38m length)	NO_3-N 200 mg/L pH 7.2	EBCT 26 minutes COD/N ratio 3 (methanol)	NO_3-N 5.7 mg/L; NO_2-N 0.02 mg/L; COD 70 mg/L	(Ergas and Rheinheimer 2004)
Pilot-scale fixed-bed bioreactors packed with sand or plastics.	ClO_4^- 77 μg/L	EBCT 15 minutes Acetic acid as a carbon source	Reduced to <4 μg/L	(Min et al. 2004)
Six-month pilot at the Castaic Lake Water Agency, Santa Clarita, CA, using fixed-bed bioreactors	18-20 mg NO_3-N/L 17-20 μg ClO_4^-/L	EBCT 15 minutes Acetic acid as a carbon source	Reduced to less than detection limit	(Brown et al. 2005)

Study Scales	Water Qualities	Parameters	Effluent Qualities, Efficiencies, and Findings	References
Bench-scale Granular Activated Carbon (GAC) bioreactor	ClO_4^- 50 μg/L	EBCT 25 minutes Acetic acid as a carbon source	Reduced to less than detection limit	(Brown et al. 2002)
Bench-scale hydrogen permeable membrane fixed film reactor	1,000 mg NO_3-N/L and/or 500 mg ClO_4^-/L	EBCT 48 hours 72 cm^2 membrane surface	80% nitrate removal and 60% ClO_4^- removal (no nitrate); the presence of nitrate reduced ClO_4^- removal.	(Chung et al. 2007)
Bench-scale hydrogen permeable membrane fixed film reactor	12.5 mg NO_3-N/L 9.4 μg ClO_4^-/L	EBCT 55 minutes 83.6 cm^2 membrane surface	99.5% reduction of 0.21 mg NO_3-N/cm^2•d and 3.4 μg ClO_4/cm^2•d	(Ziv-El and Rittmann 2009)
Pilot up-flow packed bed reactors (plastic media)	NO_3-N 325 mg/L ClO_4^- 6.37 mg/L	EBCT 15 h pH of 7.0	Completely removed perchlorate and nitrate with up to 10% salinity	(Chung et al. 2010)
Pilot anthracite filter (0.31 m deep and 1.0 mm ES)	NO_3-N 2 mg/L ClO_4^- 50 μg/L	EBCT 47 minutes	Reduced to <2 μg/L with temperature as low as 10 ºC	(Dugan et al. 2009)

Table 7. Design parameters and findings for the removal of nitrate and perchlorate in previous studies

Perchlorate occurs in water due to natural presence or manufacturing for ammonia perchlorate (Srinivasan and Sorial 2009). Being a strong oxidant, perchlorate was used as solids propellants for rockets and missiles or used for fireworks. The US EPA is not currently regulating perchlorate in drinking water but already placed it in the contaminant candidate list.

Table 7 also summarized findings and operating parameters in previous studies for pechlorate removal. Usually perchlorate is removed simultaneously with nitrate. Because the standard oxidation potential of the perchlorate/chloride pair (1.28 V) is much higher than nitrate/N_2 pair (0.75 V), perchlorate is reduced preferentially, and can be reduced down to less than 4 μg/L with an EBCT at least 15 minutes. High reduction rates of nitrate and perchlorate occurred in a synthetic high-strength salt medium 20 g/L (~2%) NaCl, while 40 g/L NaCl slowed reduction by 40% or more (Chung et al. 2010).

Similar to nitrate removal, biological processes are still cost effective for perchlorate removal. In an anoxic environment, perchlorate is reduced to chloride at the expense of an external carbon source.

4. Conclusions and perspectives

While BAFs are playing an important role in contaminant removal from water sources, understanding the process design parameters such as EBCT, media selection, backwash velocity, pH, temperature, oxygen demand, pre-oxidation requirement, inhibiting metal elements, etc, is important in that it will provide insights on treatment process control, pathogenic impacts, disinfection by-product control, and the potential to improve treatment efficiencies.

An EBCT of 10 minutes is generally recommended for the removal of TOC, MIB and geosmin, iron and manganese, arsenic, and ammonia. At least 15 minutes are generally required for the removal of nitrate and perchlorate. Backwash was found not influencing the process performance. A GAC/sand filter produced better performance than an anthracite/sand filter for the removal of NOM and taste and odor compounds, and the GAC/Sand filter out performed the anthracite/sand filter by 11% for AOC removal and showed better resistance to temperature at 1-3 ºC. Unlike the situation with NOM, placing ozonation in front of GAC filters was not benefitting the removal of MIB and geosmin. Simple aeration is sufficient for providing oxygen for the removal of ammonia, and iron and manganese.

Effective microbial adhesion and immobilization is essential for biofilm activities, and still drives further research on physicochemical (for example roughing, grafting, coating, etc) and biological (inoculated with selected species) approach in BAFs. To improve the process efficiency, dedicated microbial species targeting specific contaminants are usually desired. However, it is challenging and presents significant scientific and engineering opportunities to select microbial communities in biofilms specifically adapted to targeted contaminants. Besides currently employed media, alternative cost effective media are always interested, especially the ones from waste materials and engineered with specific surface properties.

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