A Study of Biological Water Treatment Methods For Removing Toxins From Contaminated Water Sources

*1Dr. Monalisha Mallick, Pragyan Priyadarshini,
*1Asso. Professor, Dept of Civil Engineering, NIT, BBSR,
Asst. Professor, Dept. of Civil Engineering, NIT, BBSR
Email: *1MonalishaMallick@thenalanda.com,
pragyanpriyadarshini@thenalanda.com,

A R T I C L E I N F O

A B S T R A C T

The technology of biological water treatment for the removal of toxins from water resources is the subject of the first critical examination in this study. In the future, especially for developing nations, the biological approach is thought to be the most effective way to purify drinking water. The review's primary focus is on single and combined treatment technologies that have been investigated for both surface water and ground water resources used as sources of drinking water. These treatment procedures can handle heavy metals, organic natural matter, inorganic non-metallic matter, disinfection by-products, endocrine disrupting chemicals, and microbiological pollutants in polluted drinking water sources. Given that this technique is a relatively new idea for producing safe drinking water and that there have been very few research carried out in developing nations, the possible risks and difficulties of adopting the biological process have also been explored.

1. Introduction

For people, plants, and animals, clean water is essential. In order to break down large molecules into smaller ones and move them to other regions of the body, for instance, humans need water as part of the metabolic process. Before humans can drink the water after it has been contaminated by toxic substances, it must first be disinfected. Similarly, tainted water will become dangerous for plants and animals if they consume it and expose their metabolic processes. A few important variables, including industrial and sewage effluent discharge, agricultural industry, illegal rubbish dumping, and landfill leachate leaks, cause water contamination [1]. Despite the fact that the majority of industrial firms have their own wastewater treatment facilities, if the effluent is discharged daily, the concentration of polluta UGC CARE Group-1,

nts will build up over time and increase. For drinking water treatment facilities (DWTPs), which use river water as one of their primary sources for producing safe drinking water, this phenomena will pose a significant concern in the future. At most DWTPs around the world, only traditional treatment methods including screening, coagulation/flocculation, filtration, and chlorination/fluoridation are used. High amounts of contaminants including organic carbon, nitrogen compounds, and heavy metals cannot be efficiently treated by these treatment methods. Certain DWTPs must occasionally stop operating until the level of contamination falls naturally due to dilution by rain. Because the effluents from industries can only be controlled, thus to overcome the problem a new approach has to be implemented for the continuity of clean and safe drinking water production.

The biological process started to become popular for wastewater treatment in the early 1900s [2], but not for drinking water treatment. Although biological drinking water treatment (BioDWT) has been used since the 1800s [3], its application globally is still limited. Recently, some developed countries, such as the USA, Canada, Greece, Croatia and China, have applied the BioDWT for clean and safe drinking water treatment, but there have been very limited studies performed in de- veloping countries. The BioDWT is totally dependent on the non-pa- thogenic bacteria acting as biocatalyst for biochemical oxidation, de- grading the pollutants in the contaminated drinking water and producing biologically stable water to prevent the growth of micro- organisms in the water distribution system [2].

This study aims to comprehensively review the technology of bio- logical treatment systems for safe drinking water production. This re- view will cover all types of drinking water resources, including surface water (river, lake and reservoir) and ground water that have been treated using biological technology.

Table 1

Adverse effect and limits of contaminants in drinking water.

Contaminants classification	Contaminants	Adverse effect	Regulation in drinking water (mg/L) USEPA	Malaysia
Heavy metals	Cr	● Allergic dermatitis [6]		

Hg
- Eczematous skin, allergic asthmatic reactions, gastro-enteritis, bronchial carcinomas [7]
- Neurobehavioral deficits, immune alterations, cytogenetic damage and cardiovascular toxicity [8]

Cd   Kidney damage (USEPA 2012)

< 0.1 < 0.05

< 0.002 < 0.001
< 0.005 < 0.003

Pb
- bone demineralization and increase the risk of lung cancer [9]
- Induce hypertension in adult and inhibit development of intelligence in fetus [10]

Zero < 0.01

As   hyperpigmentation and keratosis, increased risk of internal organ, skin, and lung cancers [11]

< 0.01 < 0.01

Inorganic non-metallic breath

Nitrogen compounds   An infant below six months will suffering shortness of breath and blue-baby syndrome

NH_3-N = nil NO_2-N < 1 Sulphate   - Diarrhea problems [12] < 250 < 250
NO_3-N < 10
NH_3-N < 1.5
NO_2-N = nil NO_3-N < 10

Chloride
- No health effects associated with chloride except in the special case of impaired sodium chloride metabolism [13]
- Trihalomethanes (THMs)   Cancer and adverse reproductive outcomes [14]

Disinfection by-products
< 250 < 250
< 0.08 < 1.0

Haloacetic acid (HAA)
- increased risk of intrauterine growth retardation [15]

$HAA_5 < 0.05$ Dichloroacetic acid < 0.05
Trichloroacetic acid $<$

0.1

Microbial contamination Total coliform (fecal coliform &

Escherichia coli)

• Diarrheal disease [16] < 5% Zero

---

2. Contaminants in polluted drinking water and its regulation

Heavy metals

A metal is a material in the form of an element, compound or alloy, that is typically hard, opaque, shiny, and has good thermal and elec- trical conductivity. Meanwhile, a heavy metal is a metal with a specific gravity greater than 4 or 5 [4] and have been associated with con- tamination and potential toxicity [5]. Heavy metals that are toxic are arsenic (As), copper (Cu), zinc (Zn), nickel (Ni), chromium (Cr) and plumbum (Pb). Exposure to heavy metals thorough ingestion or in- halation is dangerous because they tend to accumulate and are retained by the human body. Some of the adverse effect and limits of heavy metal contamination are listed in Table 1. Once the human body has accumulated these heavy metals, it could potentially lead to kidney problem (Cd), skin and lung cancers (As) and neurological problems (Hg).

Natural organic matter

Natural organic matter (NOM) is usually present in surface and ground water. This pollutant can also be present in soil waters. It in- cludes both humic (hydrophobic) and non-humic fractions (hydro- philic) [17] (Fig. 1). NOM can be present in two forms, i.e., particulate organic matter (POM) and dissolved organic matter (DOC). The char- acteristics of NOM depend on the source and season [18]. NOM is re- ported to not be harmful to human health, but the reaction of this matter with chlorine during the chlorination process of drinking water treatment will form disinfection by-products, such as trihalomethanes (THMs) and haloacetic acids (HAA). The classification of NOM is de- picted in Fig. 1. The hydrophobic NOM is less soluble in water, has a high molecular weight (such as humic and fulvic acids) [17], has rich aromatic carbon, and has conjugated double bonds and phenolic structures [19], while hydrophilic NOM is less reactive and contains more carbon and nitrogenous compounds [19]. In Canada, there is no

regulation for the level of humic acids in drinking water, but this

substance has been addressed in Health Canada's drinking water quality guideline on colour. It is because the presence of humic acid in water
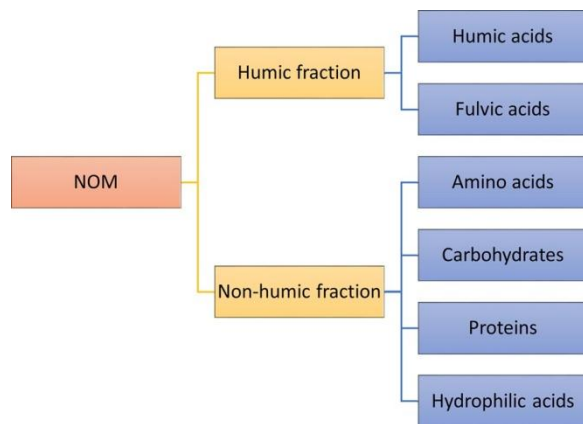
Fig. 1. NOM classification.

causes colour, odour and taste problems to increase [20].

Inorganic non-metallic matter

Inorganic non-metallic contaminants are compounds that contain no carbon or contain only carbon bound to elements other than hydrogen. Inorganic non-metallic contaminants will be discussed in this section. Inorganic non-metallic contaminants consist of nitrogen, sulfate, phosphate, and chlorides. Nitrogen compounds have been classi- fied as ammonia-nitrogen (NH_3-N), nitrite-nitrogen (NO_2-N) and nitrate-nitrogen (NO_3-N). If an infant is exposed to these nitrogen compounds in excess of the allowable limits, it could lead to the blue- baby syndrome [21]. The allowable limits for nitrogen compounds are listed in Table 1. There is no limit for the ammonia level in drinking water regulated by the US EPA. In Malaysia, the ammonia level has been regulated to below than 1.5 mg/L for both untreated and treated drinking water.

Sulfate is a chemical substance that occurs naturally in drinking water and its limit in drinking water is below than 250 mg/L. Exposure to this contaminant at high level can causes diarrhoeal diseases [12]. The presence of sulfate in drinking water at high levels may also cause
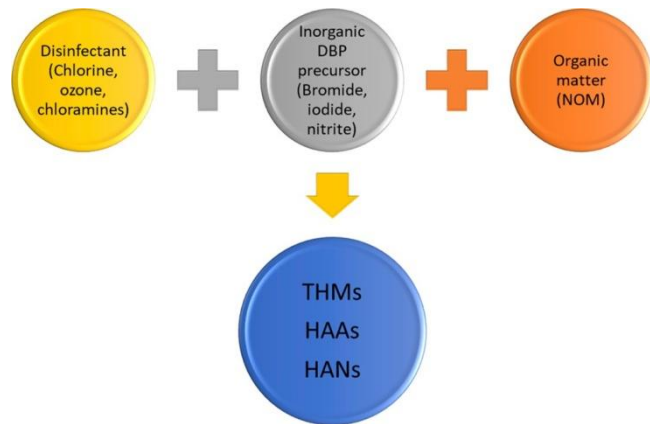
Fig. 2. The formation of DBPs.

the water to taste bitter. Sulfate can be anaerobically reduced to sulfide and then combined with hydrogen to form hydrogen sulfide gas. Once this gas accumulates in the drinking water piping system it can be corrosive due to the formation of sulfuric acid [22]. The presence of natural chloride in surface and ground water are caused by the leaching of chloride-containing rocks and soils [22]. Anthropogenic sources, such as discharge from industrial sources, landfill leachate, agricultural sources and septic tanks, also contribute to the contamination of chloride in surface and groundwater [13].

Disinfection by-products

Disinfection by-products (DBPs) are formed from the reaction of organic matters with inorganic DPBs precursors and disinfectants, such as chlorine, chloramines and chlorine dioxide, as depicted in Fig. 2. Usually, these contaminants occur during chlorination, bleaching and washing processes such as from car wash, and washing machine. DBPs can be classified into two major by-products, i.e., THMs and HAA. Haloacetonitriles (HANs) are another type of DBP that form in small amounts during the chlorination process [23]. THMs are measured as total THMs (< 0.08 mg/L), which consist of chloroform, dibromo- chloromethane, bromodichloromethane and bromoform [24]. *Meanwhile, HAAs are measured as a total of the five HAAs (< 0.06 mg/L),* i.e., the sum of monochloroacetic acid, trichloroacetic acid, dichloroacetic acid, bromate and chlorite. However, in Malaysia, only two of the most commons out of these five HAAs are monitored, i.e., trichloroacetic (< 0.1 mg/L) and dichloroacetic acid (< 0.05 mg/L). It has been re-

ported that exposure to DBPs at high levels will cause increased of risk

of cancer and adverse reproductive outcomes [14] and the risk of in- trauterine growth retardation [15].

Endocrine disrupting chemicals (EDCs)

Many of the endocrine disrupting chemicals (EDCs) may be present in surface and ground water. EDCs can be classified to four groups,

i.e., pesticides residue (DDT, methoxychlor, endosulfan), highly chlorinated compounds (polychlorinated biphenyls, dioxin, furan), alkylphenols

drinking water. This contaminant can be found in various materials, such as metals, pesticides, additives in food and personal care products. EDCs may interfere with the endocrine system, which can cause adverse reproductive health effects, including sperm count decline, hypospadias and cryptorchidism, and cancer of the breasts and testes [25]. The regulation limits for pesticides residue, such as DDT, methoxychlor and endosulfan, for the Malaysian drinking water regulation are 0.002 mg/L (DDT and methoxychlor) and 0.03 mg/L (endosulfan). The USEPA [26] has regulated that the concentration of polychlorinated biphenyls and dioxin must not higher than 0.0005 and 3×10^{-8} mg/L, respectively. This regulation indicates that EDCs in water are very strictly monitored due to their worse effects on the human endocrine system.

Microbial contamination

Contamination of microbial pathogens in drinking water causes diarrhoeal diseases. Microbial contamination includes total coliform (faecal coliform and *Escherichia Coli*), *Legionella, Giardia lamblia, Cryptosporidium* and viruses. *Contamination by these microorganisms usually comes from human and animal faecal waste that contains patho- genic microorganisms. The pathogenic microorganism must be killed or re- moved from the drinking water before it is safe to drink. The USEPA* [146] *has regulated that the total coliform in drinking must be less than 5% of the samples in a month, while Malaysian government has regulated that the microbial* contaminant must be totally removed from drinking water.

3. **Single technology of biological water treatment**

Slow and rapid sand biofiltration

Principles of SSF and RSF

Slow sand and rapid sand filters are conventionally used in drinking water treatment processing either by developed or developing coun- tries. The basic principle of a sand filter is to physically filter out the suspended solids or particles from the water. The application of a sand filter has expanded because researchers have discovered its advantages in treating water. The main advantage of a sand filter is that it not only separates the suspended solids and particles from the water but also other chemical constituents, including micropollutants [27], nitrogen compounds [28,29], heavy metals [30,31], pesticide [32], and biolo- gical contaminants (pathogenic microorganisms, such as human pa- thogenic viruses and bacteria) [33].

The outstanding performance of the sand filter in removing con-

taminants from drinking water is due to the active biological activity present at the upper layer of the sand filter, consequently upgrading the conventional sand filter system to a combination physical and biolo- gical sand filter. There are two types of biological sand filters (BSF), i.e., the slow sand filter (SSF) and the rapid sand filter (RSF). These two sand biofiltration systems have commonly been designed in various shapes either in a column, vessel, rectangular or square polyethylene water tank or a concrete tank, depending on the scale of

treatment, which can vary from a household to a full scale drinking water pro- duction plant. The major difference between these two systems are the

filtration rates. The SSF have a slower filtration rate (typically 0.1 to 0.2

and alkylphenolethoxylates (nonylphenol, octylphenol) and plastic

mh^{-1}) compared to the RSF (5-7.5 mh^{-1}) (EPA [145]). From the data

additives (bisphenol A, diethyl Phthalate, di(2-ethylhexyl) Phthalate). Among this group, DDT, methoxychlor, PCBs endosulfan, diethylhex- ylphthalate, diethylphthalate, bisphenol A, perchlorate may occur in

analysis summarized in Table 2, the size of the sand particles in the SSF was smaller (sand size range = .45–1.2 mm) compared with the RSF (sand size range 0.45–5.0 mm).

Table 2

Sand biofiltration for contaminants removal from polluted drinking water resources.

Sand system		
Treatment scale	Type of water	Type and Size of sand
Contaminants	ECBT	Filtration rate	Loading rate
Influent	Effluent	Removal (%) References

SSF Pilot plant (1.6 m X 1.6 m)

Potable water 0.29 – 0.65 mm human pathogenic viruses (bacteriophage MS2) and bacteria (*Escherichia coli* WR1)

                                                                           +

0.2-0.45 m/h –   –   –   100% Schijven et al. [33]

SSF laboratory-scale biofilter (37 cm X 15 cm)

Simulate contaminated potable water

Quartz sand (0.6–1.2 mm)

NH_4 -N 20 min   –   –   1.75 mg/L   0.93 mg/L   46.6%   Cai et al. [29]

  +

SSF laboratory-scale biofilter
65 cm x 26 cm x
26 cm)
Surface water (lake water)
– NH$_4$ -N 15 min – – 4.53 mg/L 0.08 mg/L 98.3 Hasan et al. [34]

SSF laboratory-scale (100 cm x 10 cm)
Simulate contaminate potable water
Sand (0.5 mm) NO$_3$-N 8-31 min 0.015–0.06 m/ h 27.1 g/ m2 day 22.6 mg/L 22.4 mg/L 99% Aslan and Cakici [28]

SSF laboratory scale (120 cm x 5 cm)
Simulate contaminated potable water
Red flint sand (0.71 mm)
As – 0.212 m3/m2/h – 100 µg/L. < 5 µg/L < 95 Pokhrel and Viraraghavan [30]

SSF Laboratory scale Surface water (lake water)
– 2-methylisoborneol (MIB), *trans*-1,10-
dimethyl-*trans*- 9- decalol (geosmin)
0.21 m/h MIB = ∼200 ng/L Geosmin = ∼200 ng/L
MIB = 96-62-104 ng/L
Geosmin = 20 − 26 ng/L
MIB = 48-69%
Geosmin = 87-90%
Hsieh et al. [35]

RSF Pilot scale
    (100 cm x 30 cm)
Groundwater NH_4-N, Fe and Mn   4.0 m/h   0.22 to
                        1.13 g NH4-N m^{-3} h^{-1}

NH_4-N = 0.1 ± 0.024
Fe = 0.28 ± 0.18
Mn = 0.032 ± 0.006
NH_4-N < 0.02
Fe = 0.016 ± 0.048
Mn = 0.001 ± 0.0006
Lee et al. [36]
RSF Full scale
    (7.5 × 6.2 × 3.7 m)
 Groundwater   Silica sand
        (0.9–1.0 mm)
 Fe and Mn –   8 m/h   Fe = 0.01–0.5 mg/L Mn = 0.575–
                        3.05 mg/L
 Fe = < 0.3 mg/L Mn = < 0.1 mg/L
 Fe^{2+} = < 99%
 Mn = < 96%
 Li et al. [31]
 RSF Full scale   Groundwater sand (3.0-
                        5.0 mm)
etabolites mecoprop (MCPP), bentazone, glyphosate and
p-nitrophenol
10.5 min   –   –   0.03-2.4 μg/L   Metabolites mecoprop
                        (MCPP) = 0.017-0.36 μg/L
                        Bentazone = 0.026-1.56 μg/ L
                        Glyphosate = 0.028-2.1 μg/ L
                        p-nitrophenol = 0.029-
                        2.33 μg/L
Metabolites mecoprop (MCPP) = 42-85%
Bentazone = 15-35%
Glyphosate = 7-14%
p-nitrophenol = 1-3% Hedegaard and Albrechtsen [32]
 RSF Pilot scale   Surface water Sand (0.71–1.25 mm)
  NH_4-N   –   6–7 m/h   –   0-4 mg/L (mostly 2 mg/L).   < 0.1 mg/L <
                        99%   Tranckner
                         et al. [37]
                    RSF Laboratory scale   Surface water
                        (river water)

 Sand (0.45 mm)   Ibuprofen, triclosan,
        naproxen
 15.8 2.4 m/h   Ibuprofen = 276 ± 176, triclosan, = 190 ± 42 naproxen = 170 ±
        101

4
1

buprofen = > 13.5 mg/L triclosan,= > 90% naproxen = 11% mg/L
Ibuprofen = > 95% triclosan,= > 90%
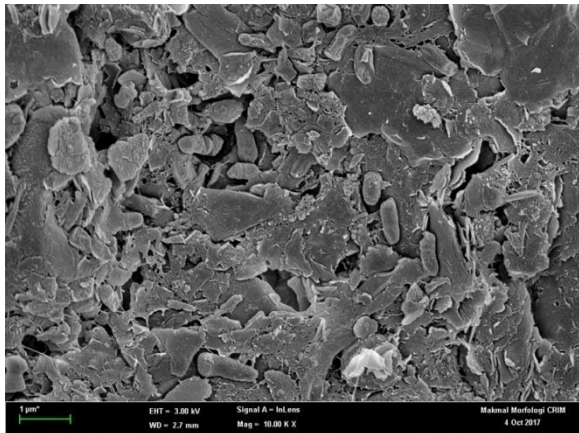naproxen = 86 ± 8.2
Zearley and Summers [27]

Fig. 3. Micrographs of a mature *Schmutzdecke* sample.

Schmutzdecke of SSF and RSF

The SSF system is based on a centuries-old bioremediation concept, where water flows slowly through a sand packing and microorganisms, or so-called biologically active *Schmutzdecke,* form and grow at the top

and bacteria [33,42–46], metals [30,31,36,47], nitrogen compounds [28,29,34,36,37,48–50], pesticide [32], pharmaceutical chemicals [27], organic chemicals (such as 2-methylisoborneol (MIB) and *trans*-1,10-di- methyl-*trans*- 9-decalol (geosmin)) [35], methyl *tert*-butyl ether (MTBE) [51], dissolved organic carbon [49,50], DBPs precursors [49,50], per-chlorate [48], NOM [52], and bisphenol [53].

Viruses and bacteria are biological pollutants that were investigated by researchers to be removed by sand biofiltration. Bacteriophage MS2 and *Escherichia coli* WR1, present in contaminated potable water, were investigated by Schijven et al. [33] to be removed using a pilot scale SSF under the various factors of temperature, filtration rate, grain sizes and *Schmutzdecke* age. The age of the *Schmutzdecke* and the grain sizes were found to contribute to the significance of the removal of bacter- iophage MS2 (0.08–3.3 $\log_{10}$ removal) and *E. coli* WR1 (0.94 with -4.5 $\log_{10}$ removal). Another study by Jenkins et al. [54] of an intermittent SSF equipped for a household could remove 0.94 log MS2 viruses (88.5%) and 1.82 log faecal coliform (98.5%) with a filtration rate of 0.01-0.03 m/h. Yahya et al. [46] achieved more than 99% removal of MS2 and PRD1. Stauber et al. [55] also reported that sampling of 55 household filters near Bonao, Dominican Republic resulted in ap-proximately 93% *E. coli* removal.

Nitrogen compounds NH_4^+-N and NO_3^--N are two famous con-

and within the sand packing [38]. The advantages of the SSF are that it is simple to operate, is an efficient and reliable system, and requires low operating and capital cost [28]. The SSF system removes the con- taminants through a few mechanisms, such as *Schmutzdecke* activity at top layer, surface catalysed degradation and mechanical filtration [28].

Fig. 3 shows a thick layer matrix of *Schmutzdecke,* containing extra-

taminants in drinking water resources. By using SSF and RSF, more than 95% of these two compounds have been successfully

removed [28,37,49,50]. However, Cai et al. [29] only achieved 46.6% of NH_4^+-N removal from simulated contaminated drinking water using an SSF system in their studies, but the removal increased, on average, to 48.5% when the

loading of NH_4^+-N increased. On the other hand, RSF would also be a good

cellular polymeric that attached to the sand surface.

alternative system for removing NH_4^+-N from groundwater, Lopato et al.

Performance of SSF and RSF

The performance of the SSF in removing contaminants depends on the water quality, surface ripening and temperature [33]. The RSF system can be designed either in open gravitational flow filters or in closed pressure filters [39]. The RSF system is operated by pouring the water in a downward flow configuration through a sand filter to remove suspended solids [40]. This system can separate more SS in a shorter period compared to the SSF [41].

The efficiency of sand biofiltration application has been studied by researchers for removing pollutants from various types of drinking water resources, such as surface water (lake and river water) and ground water, and some of researchers used simulated contaminated potable water, as summarized in Table 2. The biological systems are applied either in a la- boratory, pilot or full-scale drinking water treatment plant. From the data summarized in Table 2, SSF sand media are smaller than the RSF media, with particle sizes between 0.29 (the smallest) to 1.2 (the biggest). The types of contaminants removed by sand biofiltration are pathogenic viruses

[56] found that the removal occurred through a heterogeneous mechanism by the ammonia oxidizing bacteria. Not only nitrogen compounds but also metals, such as Fe and Mn, that are naturally present in drinking water resources can be treated using SSF and RSF systems. The RSF has been investigated by Lee et al. [36], Li et al. [31] and Štembal et al. [57] for the removal of Fe and Mn from groundwater, while the SSF system was in- vestigated to remove As [30,58] and Se [47]. At a high filtration rate of 8 m/h, the RSF system could remove more than 95% of Fe and Mn [31]. The SSF and RSF systems do not require aeration or an oxygen supply, creating anaerobic conditions at the middle and bottom parts of these systems. An anaerobic SSF has the capability to efficiently remove Se through the dissimilatory reduction of selenate to elemental selenium mechanism that is affected by the hydraulic loading rate (HLR) [47].

Biological activated carbon (BAC)

Principles and design of BAC

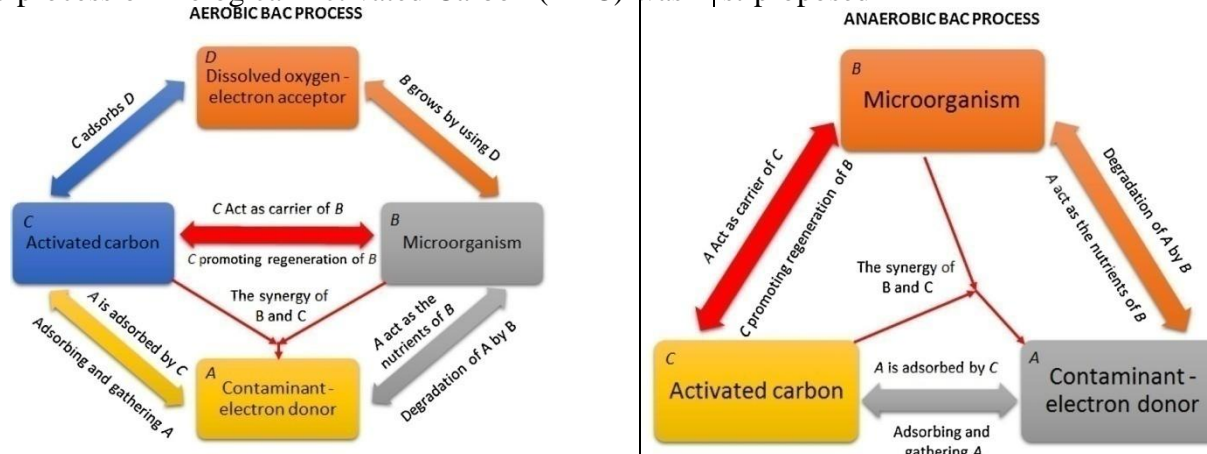The process of Biological Activated Carbon (BAC) was first proposed

Fig. 4. Illustration of the symbiotic interaction of the aerobic and anaerobic BAC processes.

by G. W. Miller and R. G. Rice in 1978, who are from the United States and Switzerland, respectively [59,60]. According to Jin et al. [59], the advantage of the microbial growing on the AC layer was first affirmed by Parkhrust and his partners in 1967. Since then, BAC technology has become an essential process for advanced water treatment in developed countries, such as America, Switzerland, Holland and Japan [59], but in developing countries, such as Malaysia, Indonesia, Russia, Turkey and others, this technology is consider as an alternative system for water treatment. In Japan, the BAC system has been used as advanced water treatment by large urban water systems since the early 1990s [61].

The BAC is developed on the activated carbon (AC) covered with a microbial, which is called biofilm. This technology combines the dual mechanisms of adsorption and biodegradation/bio-oxidation to en- hance the effectiveness of drinking water [62]. The unique character- istics of AC, such as the large surface area, ability to adsorb organic materials and the rough surface texture, are major factors of bacteria attachment [63]. The simple design of the BAC system commonly consists of activated carbon, a reactor column, feeding pump and tank.

Table (polluted drinking water treatment):

	Influent (mg/L)	Effluent	Removal efficiency	References
	59.00 ± 8.58 mg acetate-C/L			...et al [68]
	Day 0 = 3.4 mg/L Day 160 = 2.8 mg/L	Day 2.23 mg/L Day 160 = 0.79 mg/L	...%, 34.3%	Rao et al [69]
	0.15–0.98 mg/L	0.0... mg/L	Day 10 = 1.2%	...al [70]
	9012 ± 4837 ng l−1		9.9%	Zhang et al [71] developed 200... int.
	3.22 ± 0.07, 3.01 ± 0.46, 0.46 ± 0.18, 4.01	1.43	84.4% C3, 93.5% C4, 85.2% C4, 50 & 46%	Akcay et al. [72]
	0.5 mg/L, 0.02 mg/L	0.035 mg/L, 50 ng/L.	93%, > 99%	Kim et al. [73], Li et al. [62]
	–	–	Increase PACl 0 to 0.8 mg/L: 39 to 70%	McKie et al. [74]

This system does not require an aeration supply but depends on the pollutants to be removed, because some microbials, especially aerobic microbes, require dissolved oxygen (DO) from an electron acceptor for the growth and pollutants degradation. As depicted in Fig. 4, the aerobic BAC process requires DO for the growth, instead of the anae- robic process. There is symbiotic interaction between AC and micro-

organisms, where the AC acts as the carrier of microbes in the form of biofilm, while the microbes promote the regeneration of AC. The bioregeneration of the organic substrate saturated AC occurs through the mechanism of organic substrate consumption by the microorganism as a source of energy and food [64].

Activated carbon

There are various types of activated carbon that have been used worldwide for wastewater treatment processes, such as crab shells [65], papaya peels biowaste [66], coal [49,50], and sugarcane bagasse [67], as summarized in Table 3. Activated carbon is only specifically used for the BAC system, where at certain period, the AC has reached it satu- rated capacity of adsorption and during the meantime, the removal of contaminants is mainly performed by the active microbe attached on the AC surface, and only slightly by the AC. Even though some of the researchers reported that the adsorption and biodegradation of organic contaminants can occurs simultaneously, and the adsorption of AC is bioregenerated by the microorganism, the mechanisms of simultaneous adsorption, biodegradation and bioregeneration are not well identified.

Performance of BAC

The removal of various types of contaminants using the BAC system has been studied, such as nitrogenous compounds [70,71,73], organic carbon [75,68] and micropollutants [62,74]. Yapsakli et al. [70] in- vestigated the removal of NH_4^+-N using two types of activated carbon that were prepared using different methods (chemical activation; CA- gran, and heat activation; Norit 1240). In the study, the CAgran media did not adsorb NH_4^+-N at the initial stage of treatment, as Norit 1240 had done. The removal only occurred after day 12 to the acclimatiza- tion of (AOB) in the biofilter compared to the Norit 1240 where the removal occurred immediately at day 0 due to sorption effect. Another study by Kim et al.

[73] using PAC found that up to 93% of NH_4^+-N can be removed using PAC media. However, the species of microorganism community involved in the process was not investigated, only the biofilm density through heterotrophic counting plates was reported.

Recently, BAC reactors have received researcher intention for the removal of micropollutants from surface water. Li et al. [62] in- vestigated the elimination of 17β-estradiol (E2) from surface water using a BAC reactor filled with a Calgon Filtrasorb® 400 (F400) coal- based GAC media. Without seeds of any microorganism, the BAC was able to remove E2 up to 100%, and at the end of the monitoring, mi- crobial activities were visually observed to naturally develop. Mckie at

al. (2016) researched the combination of exhausted GAC-sand media in the BAC reactor with and without the inline addition of alum (0.2 and

0.8 mg/L) for the removal of spiked EDCs and PhACs. The addition of alum improved the BAC performances in removing both micro-pollutants. In the study by Mckie et al. [74], there was no microbial community reported, but the removals are believed to occur due to the microbial activities. The reason is because the age of the GAC used in the study was 4 years of operation and it was affirmed to have no ad- sorption capabilities. More advanced BAC systems were developed by Lompe et al. [76] that had mixed PAC with iron oxide nanoparticles to improve the efficiency of the BAC reactor, which consequently changed the conventional BAC to a magnetic BAC (MBAC). The iron oxide na- noparticle not only contributed to DOC and ammonia removals, but it is suitable to support the growth of nitrifying and heterotrophic bacteria.

Trickling filter (TF)

Principles and design operating of TF

A trickling filter is a fixed-film biofilter using packing medium for the biofilm attachment. The biofilm that develops on the packing medium contains microorganisms that are useful for the biodegradation of the contaminants. There are three keys component of the TF design, i.e., the rotary distributor, packing medium and air circulation. The influent of water is designed at the top of the packing medium through the rotating distribution arms. Packing mediums, such as rock or plastic, are common mediums for the conventional TF operation to treat wastewater, and there a few that have been used for drinking water treatment, such as gravel [77], TKP 312 polypropylene [78], and Silicic gravel [79], to increase the treatment efficiency. To maintain air cir- culation, an underdrain system is designed at the bottom of the TF, thus giving the advantage of not requiring an external air supply for the operation [80]. Additionally, the underdrain is useful not only for the air circulation but also for the effluent of the TF. However, some re-

searchers designed an external air supply to the TF system, such as the

study by Gouzinis et al. [77] that used a 10 L/min air flow rate.

Performance of TF

As seen in Table 4, the application of the TF for the contaminant removal from drinking water sources are not as varied compared to the wastewater. In wastewater treatment, the TF is applied for the removal of organic matter [81,82], nitrogen compounds [82–84], Cr(VI) [85], cyanide and thiocyanate [86], while for drinking water, it is favourable for the removal of NH_3-N, Fe and Mn. To the best our knowledge, there were no other contaminants from drinking water sources treated by the TF.

Groundwater treatment by full-scale TF had been studied by Tekerlekopoulou et al. [87]. In the study, under the effect of pollutant and hydraulic loading and temperature, it showed that the full-scale TF efficiently performed in removing NH_3-N, Fe and Mn. The implementation of the TF changed the lives of the New Vouprasio popu- lation, relieving them from paying for bottled water and worrying about the water supply to their home. Another full-scale TF was studied by de Vet et al. [89] for the removal of Fe from the groundwater. Mi- chalakos et al. [88] investigated the removal of Fe from potable water under a high hydraulic loading of 453-677 m^3/m^2.day. They concluded that the Fe removal using a combination of biological and physico- chemical TF was more efficient compared to the physicochemical TF alone.

The height and size of the packing medium in the TF influences the performance of the TF. Under similar hydraulic loading, Tekerlekopoulou et al. [79] found that a monolayer TF with a 1.9 mm gravel size could totally eliminated Mn from the potable water. The performance was not comparable with the multiple layer TF with packing with variable gravel sizes (up to 3.9 mm). This indicates that the performance of TF is better with a small and single size of packing material. The TF was also reported to simultaneously remove NH_3-N, Feand Mn. Additionally, as reported by Gouzinis et al. [77], Tekerleko- poulou and Vayenas [80] and Tekerlekopoulou et al. [87], the TF system could be applied for the simultaneous removal of NH_3-N, Fe and Mn. This ability will reduce the number of unit operations in elim- inating the contaminants from the contaminated drinking water.

Some researchers reported that the microbial community plays a key role in the efficient removal of contaminants by TF. In the study by de Vet et al. [89], the community of *Gallionella* sp. was closely mon- itored to ensure the bio-oxidation of Fe, but the other species commu- nities in the groundwater and TF packing media that is believed to also be responsible for the removal was not reported. Gouzinis et al. [77] also did not report about the microbial community in detail. Reported by Akker et al. [78], under a low influent NH_3-N concentration, a sig- nificant amount of the *Nitrosomonas* population was established on the TKP 312 polypropylene packing media. The lacking information or investigation on the microbial population in the biological TF system for the drinking water treatment might be due to the biosafety issue regarding pathogenic microorganisms. This issue will be discussed in detail later.

Biological aerated filter (BAF)
Principles and design operating of BAF

The biological aerated filter (BAF) is well known in wastewater treatment but not for water treatment. This system combined the at- tached and suspended growth processes. The BAF system is different from the BAC, SSF, and RSF. The term 'aerated' shows that it must be supplied an aeration for the aerobic microorganisms to grow and co- assist in the degradation and remediation of pollutants, unlike the

BAC, SSF and RSF that operated without aeration supply. The BAF can be designed in a rectangular, square or column shape, either with an up- flow or down-flow influent configuration. The design shape can vary, but this system must have an aeration configuration, backwashing system and biofilm carrier (floating or submerged media). Fig. 5 shows the schematic of the BAF for the drinking water treatment as studied by [90–92]. The shape of the BAF system will influence the oxygen transfer rate (OTR) to the biofilms, where a higher surface area will increase the OTR and vice versa. Whereas the aeration system must be efficient
enough to ensure that the DO in the BAF is at an appropriate level for
the biofilm growth and pollutants removal. An aeration system that is inefficient in supplying the air will lead to slow biofilm growth or may cause damage and, consequently, low performance in pollutants re- moval from contaminated water. The operating conditions are a crucial part of the design due to the differences in the OTR for each micro- organism for each pollutant. At a low aeration intensity, some bacteria, such as manganese oxidizing-bacteria, could perform well in removing manganese, but not the ammonia oxidizing-bacteria, which requires a higher aeration intensity [90].

   In addition, a backwash system is required to remove the excessive microbial biomass in the BAF system and to prevent the biofilm carrier from clogging. Once the water head loss in the BAF drops, the back- washing system can be operated. According to Han et al. [93], the backwashing system was operated once per 15 days in the summer and once per 7 days in the winter, while Hasan et al. [94] operated the backwashing system every two weeks, depending on the sampling and removal performance. Biofilm carriers that have been used in the BAF system are types of floating media, such as polypropylene media [95,96] and polyethylene media [97], and submerged media, such as lava [93] and zeolite [97], pebbles and ceramite [98]. Basically, sub- merged media has the adsorption capability, so the pollutants will in- itially be removed through the adsorption mechanism followed by the degradation mechanism by the biofilm. Unlike the submerged media,
the floating media just acts as a support media for the biofilm and the
removal mechanism only involves the biofilm role itself.

Table 4
Operating and performances of trickling filters.

Scale of treatment	Type of water	Media	Depth of media

Contaminants   Loading rate Influent (mg/L) Effluent (mg/L)

Removal (%)   References

Pilot scale (160 cm X 9 cm)   Potable water   Upper part: Silicic gravel (3.9 mm)

Lower part: Silicic gravel (1.9 mm)

70 cm

73 cm

Ammonia, iron and manganese

–   Iron: 0.97 mg/L

Ammonia: 0.61 mg/L

Manganese: 0.45 mg/L

–   –   Tekerlekopoulou and Vayenas [80]

Pilot scale (160 cm X 9 cm)   Potable water   Monolayer: Silicic gravel (1.9 mm)

143 cm Manganese   226 m3/m2 day.   1.02-1.98 mg/L   0100% Tekerlekopoulou et al. [79]

Multilayer

Upper part: Silicic gravel (3.9 mm)

Middle part: Silicic gravel (2.4 mm)

Lower part: Silicic gravel (1.9 mm)

26 cm

39 cm

78 cm

226 m3/m2 day. 452m3/m2 day

0.62-2.08 mg/L

1.06-2.21 mg/L

-0.25 mg/L

0.35-0.96 mg/L

88.2-100%

54-83.4%

Full scale (350 cm X 200 cm)   Groundwater   Silicic gravel (5 mm) 150 cm   Ammonia, iron and

manganese191 m3/m2 d   Ammonia: 400-550 µg/l

Iron: 150–220 µg/l

Manganese: 45–55 µg/1

Ammonia: < 25 µg/l Iron: < 2.5 µg/l Manganese: < 20 µg/l

Ammonia: > 95%

Iron: > 99%

Manganese: > 65%

Tekerlekopoulou et al. [87]

Pilot scale (400 cm X 150 cm)   Potable water   TKP 312 polypropylene Ammonia   173 Lm^{-2} d^{-1} 0.5–5.0 mg/L

< 0.2 mg/L > 95%   Akker et al. [78]

Pilot scale (180 cm x 9 cm)   Potable water   Gravel (5 mm) 160 cm   Iron   453-677 m3/m2.day

3-4 mg/L   < 0.2 mg/L   < 93%   Michalakos et al. [88]

• Manganese: Ammonia, iron and

680 m^3 m^{-2} d^{-1} Iron: 0-10 mg/L   Ammonia: 0.5 mg/L

–   Gouzinis et al. [77]

Pilot scale (180 cm x 9 cm)   Potable water   Gravel (5 mm) 160 cm

manganese

• Ammonia: 2 mg/L

mg/L

Manganese:

< 0.05 mg/L

Ammonia: 1-15 mg/L   Iron: < 0.2 mg/L

• Iron: 5 mg/L

• Manganese: 0.7 mg/L

Manganese: <

mg/L

Manganese: 0-10 mg/L   Iron: < 0.2 mg/L

• Ammonia: 2 mg/L
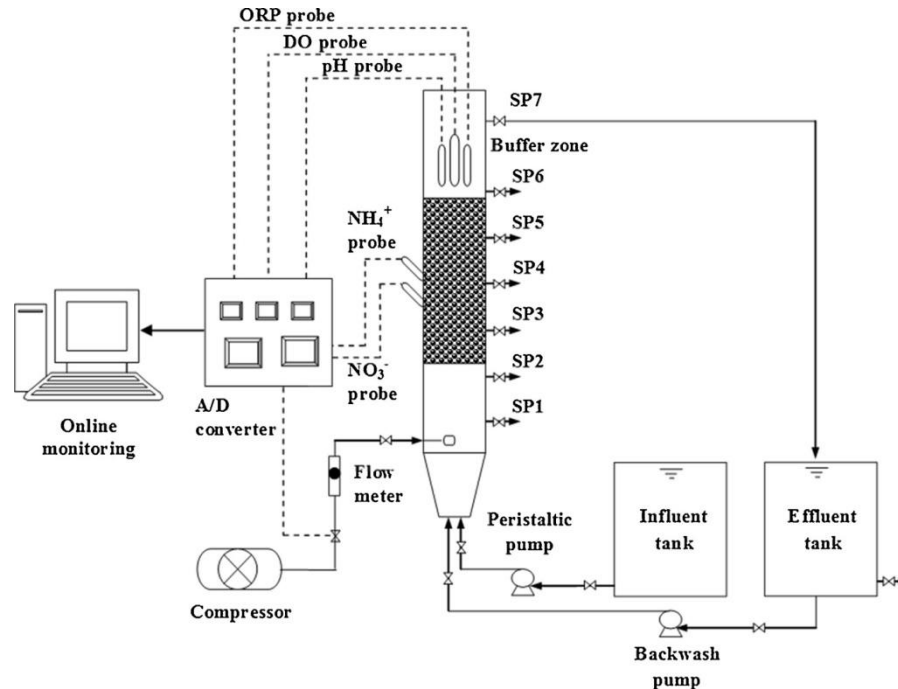• Iron: 5 mg/L

Ammonia: (2 mg/L - no reduction)

Fig. 5. A schematic of the BAF system for the drinking water treatment.

Performance of BAF system

Single treatment. The performance of the BAF system for drinking water treatment can be influenced by many operating factors, such as the aeration flow rate, hydraulic retention time or hydraulic loading, loading rate of pollutants, types of biofilm carriers, and microbial community. To the best of our knowledge, there is no investigation of the backwash effect on the BAF performance for the

have their own advantages and disadvantages to ensure the effective performance of the BAF system in drinking water treatment.

Integrated with other technology. The performance of the BAF system in treating contaminated drinking water sources can be enhanced by integrating it with other process.

To enhance the removal of DOC, NH_4^+-N and NO_2^--N by the BAF system, Han et al.

drinking water treatment, but for wastewater treatment, this factor is a priority [99–101]. Different from the other technologies, there is a lack of studies using BAF for water treatment. From the literature, few researchers investigated this system for water treatment. The researchers focused on the removal of NH_4^+-N [92,93], diclofenac [102], dissolved organic carbon [103], manganese [92,93] and ferum [104].

Previous researchers [90–92,94,95,105,106] had extensively in- vestigated the BAF

performance in the simultaneous removal of COD, NH_4^+-N and Mn^{2+}. In their study, the researchers investigated the kinetics of the simultaneous removals, the effective microbes involved, the influence of the aeration rate, and control of the DO through on-off aeration supply. Han et al. [93] applied the BAF system for the removal of NH_4^+-N and Mn^{2+} from polluted drinking water sources. The op- erating factors of the hydraulic loading and aeration intensity affecting the performance were investigated and the results showed that the hydraulic loading increased from 3.56 to 7.12 m^3/m^2.h, which had a significance effect on the removals alongside the BAF height. Mean- while, they found that a low aeration intensity had a positive effect of the Mn^{2+} removal but not for the NH_4^+-N removal due to oxygen competition between the AOB and MnOB. On the hand, Han et al. [97] also investigated the removal of NH_4^+-N and organic matters from the drinking water using a double layer BAF filled with clinoptilolite and polyethylene media compared with a single BAF using lava. They

concluded that the dual layer BAF was easier to maintain since the backwashing frequency is less compared to the single layer BAF. By using submerged media such as lava, sand, zeolite or activated carbon, the clogging problem occurs easier. It may be because the void space between the media is less, whereas floating media, such polyethylene and polypropylene media, has larger void spaces, which can reduce the potential of the clogging problem occurring. Nevertheless, both media

[103] integrated the system with a pre-oxidation process using a permanganate composite chemical (PPC). The dose of 0.6 mg/L PPC modified the structure of the DOC to the simplest structure, making it easier to be degraded in BAF system. Although the addition of PPC in the contaminated drinking water did not show a negative effect on the AOB growth [103], other pre-oxidation chemicals, such as hydrogen peroxide (H_2O_2) and ozone, must be investigated to ensure its toxicity levels towards biofilms thus prevents the negative effect on the BAF performance.

Another example of a BAF integrated with other process was the study by He et al. [102], where the researchers combined the BAF with electron beam (EB) technology (EB-BAF) for the degradation of diclo- fenac (DFC). At a 1 kGy gamma irradiation, almost 100% of the DFC was degraded to other by-products and by combining the technologies, the water quality was improved with not only the DFC degradation but also for other pollutants, such as COD and NH_4^+-N [102].

Integrating EB-BAF may be efficient for the drinking water treat- ment in small scale, but to implement it in full scale, this integration must be thoroughly investigated, including investigating the variation of pollutants in the water and the effect of radiation towards the biofilm community in the BAF system. Some question include how can the EB treat a gallon of water for continuous distribution, what is the cost of the technology and how would the increased radiation when handling this technology affect humans.

Membrane bioreactor

Principles and design operating of membrane bioreactor

The process of using the membrane bioreactor (MBR) for drinking water treatment was first proposed in the 1980s, and the first mem- brane bioreactor plant was built in France in 1988 [107]. This
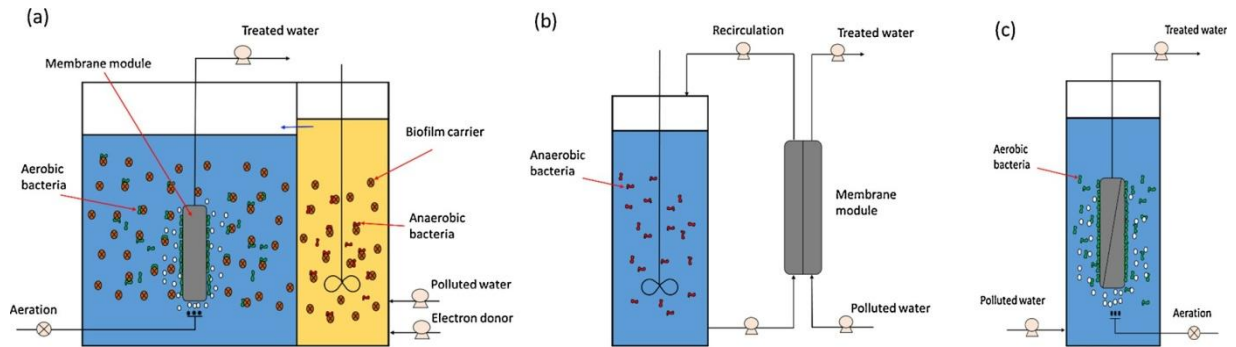
Fig. 6. Basic design of the a) AttMBR, b) IEMR, and c) SuMBR processes.

technology may overcome the problem of microbiological contamina- tion of treated water and support the growth and formation of a se- lected microorganism biomass in the reactor, which can also act slowly by removing the biodegradable refractory organic contaminants [108]. The MBR has dual functions and can be applied for the separation of the solid and liquid phases by using membranes, such as microfiltration (MF), ultrafiltration (UF), nanofiltration (NF) and reverse osmosis (RO), whereas all processes produce permeate and retentate. The separation ranges are as follows: 0.1 to 1 µm (MF), 0.005 to 0.1 µm (UF), 0.001 to 0.005 µm (NF), and 0.0001 to 0.001 µm (RO) [109]. The MF aims to remove of particles, sediment, algae, bacteria, and protozoa. While the UF membranes target the elimination of small colloids and viruses, the NF membrane targets the elimination of dissolved organic matters and divalent ions. Lastly, the RO, which is a non-porous membrane, is capable of removing monovalent ions. However, according to the lit- erature survey, most of the studies are using MF and UF for the se- paration. It is impossible to use high quality membranes such as NF and RO due to problem in the separation process where high pressure and power are required to extract clean water from the membranes. In addition, the biofilm biomass causes a fouling problem, which will also contribute to the high-pressure requirement.

The membranes used in MBR are made up of polymers, for example, polyvinyl fluoride (PVDF), polyethylene (PE), polypropylene (PP), polyethersulfone (PES), polyamide (PA), polyacrylonitrile (PAN), and poly(amide-hydrazide) (PAH). The most important characteristic of the polymer material is hydrophobicity, and all the polymers listed above are hydrophobic. Hydrophobicity will occur as the interactions between the surface of the membrane and the components present in the feed water [110]. The membrane can be a hollow-fibre or flat sheet mem- brane which is submerged in the reactor containing the bacteria bio- mass. At initial development, the MBR is developed as a suspended growth membrane bioreactor (SusMBR), but currently the reactor de- signs are innovated as an attached growth membrane bioreactor (AttMBR) and an ion exchange membrane bioreactor (IMEBR). Fig. 6 shows the schematic principles design of the MBR system.

The accumulation of bacteria biofilm at the surface of the mem- brane effects the decreasing in permeate flux or an increasing in the transmembrane pressure (TMP) during a membrane process. This loss in performance of the MBR process is called membrane fouling, which it is the result of several complex occurrences involving electrostatic and hydrophobic processes. Membrane fouling is one of the main limita- tions of the MBR process for

drinking water treatment [109,110]. Ad- ditionally, MBR technology is also facing several research challenges, such as MBR standardization, membrane life-span and operation cost [111].

Performance of MBR

The biological suspended growth reactor is combined with the membranes in the MBR to act as a separation device to solve some of the problems, such as denitrification of nitrate [108]; [112–115], am- monia removal [116] and reduction of total organic carbon (TOC) [116]. Form the literature survey, it was found that most of the re- searchers used MBR for the removal of nitrate. Ravnjak et al. [114] achieved high nitrate removal from polluted groundwater by using a two-stage membrane bioreactor (anoxic and oxic). In the study, a PE membrane with a pore size of 0.4 μm combined with a Biochip carrier was used in oxic, but in the anoxic zone, only Biocontact-N carriers were used to enhance the nitrate removal. Applying a biofilm carrier in the MBR would enhance the contact area between the biofilm and the pollutants compared to the conventional MBR that only applies the suspended growth processes. However, this innovation showed a good nitrate removal, but maintaining both biofilm carriers in the anoxic and oxic zone is difficult, and over a long period, the carrier will settle to the bottom tank due to the high density of the dead biomass. Moreover, in the study by Ravnjak et al. [114], the membrane zone was located in an oxic zone, which would increase the fouling problem and shorten the membrane lifetime. The fouling can be controlled through.

As reported by McAdam and Judd [113], the MBR can removed high nitrate in the water by manipulating the C/N ratio using different carbon sources, such as methanol and ethanol, for the biofilm meta- bolism. In addition to the nitrate removal, manipulating the C/N ratio could also increase the hydrogen production during the treatment, which can be beneficial for the DWT operation. Even though the membrane could easily be fouled due to biofilm attachment on its surface, for a long-term operation, it can degrade the high molecule organic compounds that are retained in the MBR, which consequently prevents the formation of THMs in the treated water. Some researchers have investigated the technique by embedding nanoparticle on the membrane surface to keep the membrane from fouling. Although it shows good performance in controlling the fouling, the embedding process requires a sticky procedure and is difficult and requires a high cost for full scale production.

To obtain high-quality drinking water, conventional MBR has been innovated through the application of ion exchange MBR (IEMBR). The IEMBR has good potential not only for macropollutants removal but also for anionic micropollutants, such as perchlorate [117] and bromate [118]. As an example, a study by Ricardo et al. [117] uses IEMBR for the removal of nitrate and perchlorate from drinking water. The biofilm attached on the membrane surface plays an important role for both pollutants reduction. In the study, heterotrophic denitrifiers biofilm take the first action to remove nitrate. Then, it is followed by the per- chlorate reducing biofilm, which removes the perchlorate. To ensure that the IEMBR performs well, an electron donor, such as ethanol, must be added to the reactor for the biofilm metabolism. However, not controlling the addition of the electron donor would increase the acetic acid production, which will reduce the pH and provide unsuitable conditions for biofilm growth.

Other biofiltration system

Moving bed biofilm reactor

The moving bed biofilm reactor (MBBR) is a type of wastewater

treatment process that was first invented by Hallvard Ødegaard at Norwegian University of Science and Technology in 1989. MBBR is well known in wastewater treatment but there has been very limited study on its use for drinking water treatment. MBBR was developed based on the concept of combining the conventional activated sludge process and the biofilm process. This technology is a completely mixed and con- tinuous operation, and it can withstand a high density of biomass but omit clogs in the reactor and sludge recycling [119]. The biological treatment process is taking place when the microorganisms that are attached at the carrier element will be in moving their position within the MBBR system [120,121].

Aerobic MBBRs use a diffused aeration system to uniformly move the biofilm carriers and meet the oxygen requirements. While operated in the anaerobic condition, the carrier movements are distributed by mechanical mixers [122]. Rodgers and Zhan [123] stated that the biofilm thickness not only can be controlled by air flow but also by mechanical mixing energy. The diffused aeration will be provided from the bottom of the MBBR and the inflow of water is coming from the top of the MBBR [124]. The increasing biomass concentration inside the MBBR was developed by increasing the amount of supporting media and lowering the head loss [144].

The carrier element in the MBBR process will preferably be made up of material that is less dense than water, such as polyethylene, poly- propylene, polymer foam pads, polyvinyl alcohol gel, polyurethane sponge and granular activated carbon. McQuarrie and Boltz [122] found that the high density of plastic biofilm carriers (0.98 g/cm^3) re- duces the carrier's tendency to float and so power is needed to uni- formly move the carriers. However, these carriers will slightly float with densities between 0.94 and 0.96 g/cm^3. In addition, they also stated that a plastic biofilm carrier in the MBBR should increase the volume up to 67% of the empty bed liquid volume.

MBBR may be made up as a single reactor or as several reactors-in- series. Normally, MBBR is designed with length-to-width ratio (L:W) in the range of 0.5:1 to 1.5:1. However, if the MMBR design is greater than that, non-uniform free moving plastic biofilm carriers will be dis- tributed throughout the bioreactor. As a result, the oxygen transfer efficiency and hydraulic capacity of the plastic biofilm carrier retention screens are reduced [122]. The performance of the MBBR technology is based on several parameters that need to be considered. They are the aeration system [122,123,125], biofilm [126], flow configuration [127], types and characteristics of the media [122], loading rate [124] and filling fraction [128].

This simple and strong biological treatment process is suitable for the treatment of nitrogen compounds from polluted raw water, but currently, there are no other types of contaminants has been studied to be removed from any type of drinking water resources using this technology. Some of the studies, such as the investigation by Zhang et al. [129], focused on the responsibility of the nitrifying community on the nitrogen compounds removal. The lack of interest by researchers to use MBBR for drinking water treatment may be due to the additional requirement of unit processes, such as floatation or clarifier for the biomass separation and the high percentage of microbial contamination in the treated water due to high microbial biomass amounts passing thorough the MBBR to the next process.

Fluidized bed biofilm reactor

Biological drinking water treatment using a Fluidized Bed Biofilm Reactor (FBBR) was studied 30 decades ago by Kurt et al. [130] for the removal of nitrate. The basic concept of FBBR technology is that the contaminated water is pumped up-flow through the biological bed at a sufficient velocity to fluidize the granular media bed. Currently, there are very few reports of

using FBBR in drinking water treatment. This may be related to the design and operation of the FBBR, where the

carriers used in the reactor must be in the fluidized condition. This condition can only be achieved by manipulating the influent flow to a certain velocity. A high velocity of influent flow rate would result in a biomass wash out from the FBBR. However, some of the advantages of this FBBR process are the non-cloggable biofilm reactor where back- washing is not required [131,132] and it is easy to manage [131].

The types of contaminants removed from drinking water sources by FBBR were TOC, THM and ammonia, which were investigated by Xie et al. [132], while Burghate and Ingole [131] investigated the removal of nitrate. Based on Xie et al. [132], the reduction of TOC was low, at only 12.6%, while the reduction of THMFP and HAAFP was significant, and the ammonia was reduced 30% to 40% (temperature below 3°C), however, the reduction rose to over 50% when the temperature in- creased above 3°C. On the other hand, the maximum removal efficiency for nitrate was 91% at an HRT of 30 min and optimum removal effi- ciency was 86% at an HRT of 10 min [131]. In the study conducted by Xie et al. [132], *Nitrosomonas* and *Nitrobacter* were used for biological nitrification. Both groups of bacteria are autotrophic, and they use carbon dioxide as the carbon source for biosynthesis and the oxidation of nitrogen compounds as the energy source.

4. **Integrated/combining technologies**

Integrating two different processes is a wise action for upgrading the drinking water treatment technology to ensure that the treated water meets the standard discharge limits as regulated by the legislation. Integrated systems for the BioDWT include the membrane coagulation bioreactor (MCBR). The BAC-MBR has been studied by researchers for obtaining the highest quality of treated drinking water.

Membrane coagulation bioreactor

The membrane coagulation bioreactor (MCBR) is a combination of the coagulation and membrane filtration process and has been suc- cessfully applied to drinking water treatment [133,134]. Liu et al. [135] and Tian et al. [136] stated that the coagulation process is the most effective pretreatment technology for the mitigation of membrane fouling, prolong the membrane life span and the improving the permeate quality. By using the MCBR process, the conventional treat- ment plant units, i.e., coagulation, flocculation, sedimentation and fil- tration, can be eliminated from the treatment process. In MCBR tech- nology, the performance is dependent on the upward internal circulation flow of water in the membrane separation region that is induced by air lift causing the water in the reactor and the feed water into the reactor to be mixed and flocculated. When the water in reactor is flocculated continuously, this will make the fine particles become larger and mitigate the membrane fouling. This is because smaller

particles easily deposit onto the membrane surface, which is the main

factor of membrane fouling due to blocking the membrane pores.

The MCBR has performed well in the removal of microorganisms and particulates, denitrification and phosphate removal. Tian et al.

[136] found that MCBR achieved much higher removal efficiencies of organic matter, such as total organic carbon (TOC) and dissolved or- ganic carbon (DOC), compared to MBR, which is caused by poly- aluminium chloride (PACl) coagulation in the bioreactor. In addition, this MCBR process effectively removed the biodegradable dissolved organic carbon (BDOC) and assimilable organic carbon (AOC), which were, respectively, 8.2% and 10.1% higher than with the

MBR. There is still a lack of research on this technology for the BioDWT. Some sug- gestion for future research are to investigate the effect of the coagulant types on the performance. Using organic coagulants, such chitosan, would provide a benefit to the performance not only for the treatment, but it can also be a nutrient for the biofilm growth in the system.

Moreover, the type of membrane can also be considered for good water quality production. According to another study by Tian et al. [137], the MCBR can be modified by adding an adsorbent, such PAC, to enhance the removal of organic matter, but this type of technology may require extensive maintenance and a process to separate the PAC.

Biological activated carbon-membrane bioreactor

The combination of the Biological Activated Carbon and Membrane Bioreactor (BAC-MBR) has been evaluated for drinking water treatment by several researchers [134,138,139]. The design setup of the BAC- MBR can be seen in Fig. 7. The combination of the BAC to MBR not only improves the quality of the treated water, but the BAC could mitigate the membrane fouling issues. Even though this type of hybrid system is easier to maintain, this technology is still receiving the researchers at- tention compared to the MCABR, where only one unit is required for two functions.

In combining the technologies of BAC-MBR, Tian et al. [134] con- cluded that the BAC was capable of enhancing the removal of high amounts of organic matter and ammonium by the adsorption and bio- degradation process, as well as mitigating the membrane fouling in the downstream MBR. Conversely, the MBR could eliminate organic matter and ammonium in the biodegradation process, as well as separating particles in the BAC effluent. Moreover, Tian et al. [134] studied one of the fundamental parameters of water quality in drinking water treat- ment, which is turbidity, which represents the particles in water. In- itially, the turbidity in the raw water was 1.88 ± 0.62 NTU, on average. As a result of the BAC performance, the removal efficiency was

$59.7 \pm 12.7\%$, with 0.70 ± 0.16 NTU still remaining in the effluent.

However, the MBR exhibited an excellent turbidity removal capacity, which is 0.06 ± 0.02 NTU through the separation of the membrane. In operating the BAC and MBR, the microbial community can be different, focusing on the degradation of specific groups of contaminants, thus it can improve the treated water quality where it is difficult to achieve at a high level in only a one-unit system.

5. Challenges in biological drinking water treatment

Drinking water is one of Earth's most precious resources. Increased demand for high-quality and safe drinking water production has led biological treatment technology to fulfil this demand. Webber [140] stated that there are many questions about the suitability, capacity, reliability, and safety of using this biological water treatment tech- nology. However, there are some threats to the drinking water sources that should not be underestimated. One of them is the anthropogenic activities in industrial areas that caused the quantity and quality of water available for human consumption to decline. Consequently, based on a review by Benner et al. [141], they found 133 micro- pollutants (i.e., pesticides and pharmaceuticals) in both the raw and finished drinking water. Not only micropollutants, some other major contaminants, such as heavy metals, organic and inorganic matters and nitrogen compounds, must be eliminated from the drinking water.

In this technology, microorganisms are used to catalyse biochemical

oxidation or reduction of drinking water contaminants and to produce biologically stable water [3].

However, the uncontrolled growth of microorganisms can occur during the drinking water treatment and distribution. This issue can lead to hygiene problems, such as the de- velopment of opportunistic pathogens, deterioration in taste, smell, and colour, or the biological corrosion of pipes [142]. A better under- standing of the interaction of bacteria in the distribution system and the environmental conditions is necessary to better control the bacteria during the treatment and distribution of drinking water.

In this review, a few challenges of the BioDWT were highlighted, as summarized in Fig. 8. The treating of drinking water biologically may cause exposure to pathogenic microorganism contaminations, such as *E. coli*. The diversity of pathogenic and non-pathogenic microorganisms is difficult to control in biological treatment due to the factors of the raw water characteristic, which always vary and contribute to the micro- organism growth in the BioDWT. An assessment of microorganism in the *Schmutzdecke* of the biosand filter for the treatment of river water showed that pathogen strains are a dominating group in the *Schmutz- decke* layer compared to the non-pathogens [143]. Introducing the
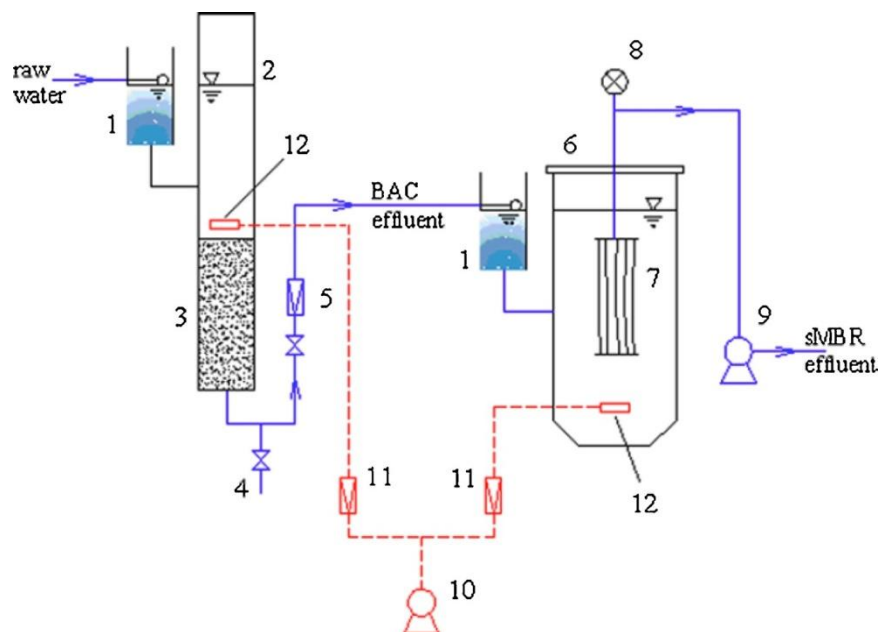

Fig. 7. A schematic diagram of the BAC-MBR system (1) constant level water tank; (2) BAC; (3) granular activated carbon layer; (4) backwashing valve; (5) BAC effluent flowmeter; (6) MBR; (7) UF membrane module; (8) manometer; (9) suction pump; (10) air blower; (11) air flowmeter; (12) air diffuser [134].
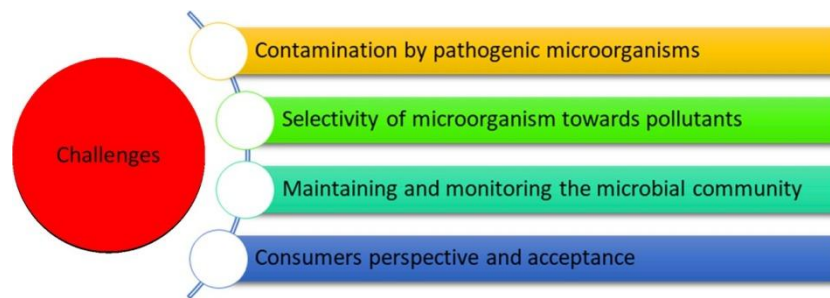
Fig. 8. Challenges in applying the biological
process for water treatment.

effective non-pathogenic microorganism in the start-up may prevent the dominating of the pathogens present in the *Schmutzdecke* or biofilm. It is difficult to achieve good pollutants removal simultaneously in a one-unit treatment system. The microorganisms in the reactor must be diverse to ensure all pollutants in the raw water can be treated. Selectivity of microorganisms towards specific contaminants may overcome the inefficiency of BioDWT for raw water treatment. As an example, the community of *Nitrospira* and *Nitrosomonas* has been re-ported to be only capable of treating nitrogen compounds, but to treat other pollutants, such as heavy metals, it requires other types of mi-croorganism groups. However, a specific genus of microorganism has the ability to remove various types of pollutants, as an example, genus *Sphingomona*s is a good choice for the removal of terpene 2-methyli-soborneol (MIB), soproturon, polycyclic aromatic hydrocarbons (PAHs)

and lindane. Thus, before start-up, the BioDWT, selection of specific microbials or general microbials as a seeding is important to ensure the stability and good performance of the reactor.

During the treatment, the community of the microbial will shift according to the times due to a few factors, such as the quality of raw water and the operating conditions of treatment, such as the pH, HRT, and the organic loading rate. Maintaining and monitoring the microbial community may pose challenges to maintaining the performance sta- bility. As reported by Liao et al. [69], at the early stage of treatment, the microbial identification only achieved at class level, but after operating for 160 days, the identification was at genus level, where its shows the microbial community change within the treatment period. This condi- tion may affect the bioreactor performance of either low performance or good performance.

The last challenge of the biological treatment to be applied for drinking water treatment is the consumer perspective and acceptance of the microbial contamination in the treated water. Consumers may be worried about the microbial contamination from the biological process itself in the treated drinking water. Without a proper explanation or information, it is very difficult to convince consumers of the safeness of the BioDWT. Before the BioDWT can applied in the water treatment, a survey on the consumer perspective and acceptance may conducted, then information on this BioDWT should be distributed to public.

6. **Conclusion**

The biological process via biofilms is a relatively new concept and is considered a 'future technology' for the production of high-quality and safe drinking water. A few technologies for contaminant removal from the level of micro up to the macro level have been discussed in this review paper. Even though these technologies are well known in de- veloped countries, in most

developing countries, the technology is still unacceptable by the water treatment practiser and consumer. Some of the technologies that have good potential for application in water treatment plants are SSF, RSF, BAC and MBR or combinations of these processes. These technologies can efficiently remove ammonia, nitrate, nitrite, organic carbon, heavy metals and various type of micro- pollutants. evertheless, information on these technologies must r treatment practiser and consumer, so that it can be accepted to be applied in water treatment plants in developing countries.

Declaration of Competing Interests

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influ- ence the work reported in this paper.

References

[1] H.A. Hasan, S.R.S. Abdullah, N.T. Kofli, S.Y. Yeoh, Interaction of environmental factors on simultaneous biosorption of lead and manganese ions by locally isolated Bacillus cereus, J. Ind. Eng. Chem. 37 (2016) 295–305.

[2] Carollo, Drinking Water Biological Treatment, (2014) (accessed 20 May 2015), http://www.carollo.com/sites/default/files/docs/BiologicalTreatment.pdf.

[3] P.J. Evans, E.M. Opitz, P.A. Daniel, C.R. Schulz, Biological Drinking Water Treatment Perceptions and Actual Experiences in North America, Water Research Foundation and Department of Defense, Colorado, USA, 2010, p. 29.

[4] A. Adal, S.W. Wiener, Heavy Metal Toxicity, (2015) (accessed 14 July 2015), http://emedicine.medscape.com/article/814960-overview.

[5] J.H. Duffus, "Heavy metals"—a meaningless term? Pure Appl. Chem. 74 (2002) 793–807.

[6] USEPA, Chromium in Drinking Water, United States Environmental Protection Agency, 2018 (Accessed on 17 January 2018), https://www.epa.gov/dwstandardsregulations/chromium-drinking-water.

[7] F. Baruthio, Toxic effects of chromium and its compounds, Biol. Trace Elem. Res. 32 (1992) 145–153.

[8] C.J. Passos, D. Mergler, Human mercury exposure and adverse health effects in the Amazon: a review, Cad. Saude Publica 24 (2008) 503–520.

[9] A. Bernard, Cadmium & its adverse effects on human health, Indian J. Med. Res. 128 (2008) 557–564.

[10] T. Agusa, T. Kunito, K. Ramu, C. Chamnan, P.T.K. Trang, A. Minh Subramanian, H. Iwata, P.H. Viet, T.S. Tana, S. Tanabe, Lead contamination and its human health effects in India, Vietnam and Cambodia, Biomed. Res. Trace Elem. 17 (2006) 413–416.

[11] S. Kapaj, H. Peterson, K. Liber, P. Bhattacharya, Human health effects from chronic arsenic poisoning–a review, J. Environ. Sci. Health A. Tox. Subst. Environ. Eng. 41 (2006) 2399–2428.

[12] L.C. Backer, Assessing the acute gastrointestinal effects of ingesting naturally occurring, high levels of sulfate in drinking water, Crit. Rev. Clin. Lab. Sci. 37 (2000) 389–400.

[13] WHO, Chloride in drinking Water, 2nd ed., Guidelines for Drinking-Water Quality Vol. 2 World Health Organization, Geneva, 1996 Health criteria and other supporting information.

[14] E. Hood, Tap water and trihalomethanes: flow of concerns continues environ, Health Perspect. Environews Science Selections 113 (2005) 474.

[15] C.K. Porter, S.D. Putnam, K.L. Hunting, M.R. Riddle, The effect of trihalomethane and haloacetic acid exposure on fetal growth in a Maryland County, Am. J. Epidemiol. 162 (2005) 334–344.

[16] J. Fawell, M.J. Nieuwenhuijsen, Contaminants in drinking water, Br. Med. Bull. 68 (2003) 199–208.

[17] I. García, Removal of Natural Organic Matter to Reduce the Presence of Trihalomethanes in Drinking Water. Doctoral Thesis, School of Chemical Science and Engineering, Royal Institute of Technology, Stockholm, Sweden, 2011.

[18] R. Fabris, C.W.K. Chow, M. Drikas, B. Eikebrokk, Comparison of NOM character in selected Australian and Norwegian drinking waters, Water Res. 42 (2008)