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Per- and Polyfluoroalkyl Substances (PFAS) in Drinking Water

Part II: Conventional and Emerging Drinking Water Treatment Technologies

Brian J. Yates, P.E., Burgess & Niple



INTRODUCTION

In the previous issue of the AWWA Ohio Section Newsletter, a primer on PFAS was published as Part I. Part II reviews conventional full-scale technologies for PFAS treatment and surveys emerging treatment technologies.

CONVENTIONAL WATER TREATMENT TECHNOLOGIES FOR PFAS REMOVAL

INEFFECTIVE CONVENTIONAL TREATMENT TECHNOLOGIES

In the absence of federal drinking water standards for PFAS, many states have enacted their own enforceable standards (see Figure 1). Therefore, PFAS treatment at full-scale water treatment plants (WTPs) is currently practiced and performance data exist. Most systems remove PFAS by sorption or separation. No full-scale system for the destruction of PFAS has been constructed, and current technologies only transfer PFAS to another medium or a concentrated waste stream which then requires further treatment and/or disposal. These systems are interim solutions: PFAS contamination can only be effectively and sustainably addressed using destructive technologies. Due to the complexity of PFAS mixtures in raw waters, it is the author's opinion that a fully-effective treatment system will be a combination of treatment technologies designed as a treatment-train.

Figure 1. States with Current and Proposed PFAS Regulation in Water

States with Current and Proposed PFAS Regulations in Water – October 2018

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STATE - First Year of Proposed Rules PFAS Regulated Matrices Regulated PFAS Concentration Range (ppb)

ALABAMA - 2016 PFOS, PFOA **Drinking Water** 0.07 ppb

ALASKA - 2016 PFOA, PFOS, PFBS, PFNA, PFHxS, PFHpA Drinking Water. Groundwater and Surface Water/ Effluent 0.07 ppb - 2 ppb

ARIZONA - 2017 PFOS, PFOA **Drinking Water** 0.07 ppb

CALIFORNIA - 2018 PFOA, PFOS Drinking Water and Groundwater 0.013 ppb - 0.07 ppb

CONNECTICUT - 2016 PEOA, PEOS, PEHXA, PFHpA, PFNA Groundwater 0.07 ppb

COLORADO - 2017 PFOA, PFOS **Drinking Water** 0.07 ppb DELAWARE - 2016 PFOA, PFOS, PFBS Groundwater 0.07 ppb - 38 ppb IOWA - 2016

PFOA, PFOS Groundwater 0.07 ppb - 1 ppb

MAINE - 2016 PFOA, PFOS Drinking Water, Groundwater and Recreational Water 0.05 ppb - 1.2 ppb MASSACHUSETTS - 2017 PFOS, PFOA, PFBS,

PFHxS, PFHpA, PFNA **Drinking Water** 0.07 ppb - 2 ppb

PFOS, PFOA Drinking Water, Groundwater, Surface Water/Effluent 0.011 ppb - 0.42 ppb MINNESOTA - 2017 PFOA, PFOS, PFBA, PFBS PFHxS Drinking Water and Groundwater 0.027 ppb - 7 ppb NEVADA - 2015 PFOA, PFOS, PFBS Drinking Water 0.667 ppb - 667 ppb **NEW HAMPSHIRE** - 2016

MICHIGAN - 2015

PFOA, PFOS Groundwater 0.07 ppb NEW JERSEY - 2015 PFOA, PFOS, PFNA Drinking Water and Groundwater

0.013 ppb -0.014 ppb NORTH CAROLINA - 2006 PEOA GenX Drinking Water and Groundwater

0.14 ppb - 2 ppb

PFHpA, PFOSA, PFNA, PFDA, PFDS, PFUnA, PFDoA, PFTrDA, PFTeDA Groundwater 0.093 ppb - 71 ppb

0.07 ppb

0.07 ppb

TEXAS - 2017

PFOA, PFOS, PFBA, PFBS,

PEPeA, PEHxA, PEHxS,



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PFOA = perfluorooctanoic acid (C8) PEOS = perfluorooctane sulfonic acid (C8) PFBA = perfluorobutyric acid (C4) PFBS perfluorobutane sulfonic acid (C4) PFPeA = perfluoropentanoic acid (C5) perfluorohexanoic acid (C6) PFHxA =perfluorohexane sulfonic acid (C6) PFHxS = PFHpA = perrfluoroheptanoic acid (C7) PFOSA = perfluorooctane sulfonamide (C8) PFNA = perfluorononanoic acid (C9) PFDA = perfluorodecanoic acid (C10) PFDS = perfluorodecane sulfonate (10) PFUnA = perfluoroundecanoic acid (C11) PFDoA = perfluorododecanoic acid (C12) PFTrDA = perfluorotridecanoic acid (C13) PFTeDA = perfluorotetradecanoic acid (C14) Gen-X = ammonium hexafluoropropylene

oxide-dimer acid

Data are from ITRC https://pfas-1.itrcweb.org/fact-sheets/ accessed October 30, 2018

The following are data from several full-scale WTPs surveyed for select PFAS in raw water, finished water, and after individual treatment technologies. Figure 2 is a schematic representation of PFAS fate in conventional WTPs. None of these systems were designed for PFAS removal, nevertheless some of the treatment technologies proved to be at least partially effective in the removal of PFAS.

Rahman, et al. (2014) present results of sampling 1. campaigns at nine WTPs. Raw water sources included groundwater, surface water, and treated wastewater. PFAS detected within raw and finished water, and after individual treatment technologies include six PFAA with four, six and eight carbons (PFOS, PFOA, PFHxA, PFHxS, PFBA, and PFBS - see Figure 1) from 0.4 to 182 ng/L. Treatment technologies included coagulation/ flocculation/sedimentation (C/F/S), slow and rapid sand filtration, dissolved air floatation (DAF), GAC, RO, ozonation, ultraviolet (UV) disinfection, chlorination, and chloramination. Influent and effluent concentrations of the PFAA

analyzed were similar at all WTPs, indicating minimal removal of PFAA. In several instances the PFAA concentrations in the finished water were higher than in the raw water, attributable to transformation of PFAS precursors, desorption from overrun GAC units, and leaching of Teflon®-coated components. Except for RO, conventional treatment technologies were unable to adequately remove the PFAA analyzed.

2. Appleman, et al. (2014) report a survey of 15 WTPs in which 23 PFAS were analyzed in raw and finished water, and after individual treatment technologies. Raw water included groundwater and surface water impacted by upstream wastewater effluent. Thirteen PFAA and three PFAS precursors were analyzed. Treatment technologies included aeration, C/F/S, DAF, ozonation, permanganate and UV/ hydrogen peroxide (H2O2) advanced oxidation process (AOP), GAC, IX, UV disinfection, and softening. Results from riverbank filtration were inconclusive. The most commonly detected raw water PFAS were PFOS. PFHxS. and PFHxA, however raw waters contained additional PFAS in complex mixtures. The highest raw-water concentrations were 370 ng/L (PFPeA) and 220 ng/L (PFOA). Under the current United States Environmental Protection Agency (USEPA) lifetime health advisory (LHA) of 70 ng/L (combined PFOS and PFOA), PFOA would be considered quite elevated. Only four years ago, this would have been less than the USEPA provisional health advisory of 400 ng/L. This illustrates how guickly the regulatory landscape is changing and how even seemingly "small" concentrations of PFAS are now a regulatory concern. Except for GAC, IX, and RO, conventional treatment technologies were unable to adequately remove the PFAS analyzed.

3. Eschauzier, et al. (2012) report the removal of select PFAS from a 50 million gallon per day

WTP in the Netherlands. Raw water was from the River Rhine, contaminated by an upstream industrial facility in Germany, illustrating how approaches to PFAS source reduction must be consistent between facilities with shared aquatic resources. PFAS analyzed included PFBS, PFBA, PFPeA, PFHxS, PFHxA, PFOS, PFOA, and PFDA. The highest PFAS concentration in the raw water was 52 ng/L. Treatment technologies within the surveyed WTP included C/F/S, aeration, ozonation, softening, and GAC. Except for RO, conventional treatment technologies were unable to substantially remove the PFAS analyzed.

 Glover, et al. (2018) surveyed four pilot potablereuse plants from different areas of the U.S. This study reported the relative removal efficiencies of ten PFAA (PFBA, PFBS, PFPnA, PFHxA, PFHxS, PFHpA, PFOA, PFOS, PFNA, PFDA), three specific PFAS precursors (N-MeFOSAA, 6:2 FtS, and 8:2 FtS), and non-targeted PFAS precursors by the Total Organic Precursor Assay (TOP Assay). Raw water PFAS ranged from 52 to 227 ng/L. The pilot plants employed ozone, biological activated carbon (BAC), GAC, microfiltration, ultrafiltration, RO, and AOPs based on UV (i.e., UV/ H2O2 and UV/Chlorine [UV/CI]). Except for GAC, conventional treatment technologies were unable to substantially remove the PFAS analyzed.

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Figure 2. Schematic Representation of PFAS Fate in Conventional Water Treatment Systems

PARTIALLY EFFECTIVE CONVENTIONAL TREATMENT TECHNOLOGIES

Granular Activated Carbon (GAC)

GAC is an established treatment technology for the removal of natural organic matter (NOM), taste and odor compounds, and organic micropollutants. GAC's ability to sorb PFAS makes it a current best management practice. GAC has been identified by the New Jersey Drinking Water Quality Institute Treatment Subcommittee as one of the most effective treatment options for removal of PFNA, PFOS and PFOA (Cummings, et al., 2015). The best proof of this is that PFAS blood serum concentrations in affected populations of the Mid-Ohio Valley decreased by 28% within one year of the installation of GAC filters (Bartell, et al., 2010).

There are four steps for sorption of contaminants to GAC (Qiu, 2007):

- 1. Diffusion of the contaminant in the liquid phase;
- 2. Contaminant mass transfer to the solid phase (GAC surface);
- 3. Internal diffusion (pore and surface diffusion within/on GAC);
- 4. Adsorption by electrostatic and/or hydrophobic interactions.



Each step presents a kinetic and/or thermodynamic barrier for the removal of PFAS. Individual PFAS will be removed by GAC with different efficiencies. Because one of the dominant sorption processes is electrostatic, the head of various PFAA will associate to different degrees with GAC. In batch adsorption tests with sediments, Higgins and Luthy (2006) showed that the Langmuir absorption coefficient for perfluoroalkyl sulfonates (PFSA) averaged 0.23 log units greater than perfluoroalkyl carboxylates (PFCA); sulfonated PFAA adsorb better than carboxylated PFAA. This is also observed at full-scale (Appleman, et al., 2014; Rahman, et al., 2014). Because the other dominant sorption process is hydrophobic, the perfluoroalkyl chain length also impacts removal; longer-chain compounds are removed with better efficiencies than shorter-chain compounds. Higgins and Luthy (2006) observed Langmuir adsorption coefficients decrease by 0.50 to 0.60 log units with each fewer -CF2- group in the PFAS molecule. Both sorption behaviors are confirmed in other studies on activated carbon where Freundlich isotherm constants followed PFBS < PFOA < PFOS (Ochoa-Herrera and Sierra-Alvarez, 2008; Hansen, et al., 2009).

PFAS sorption behavior on GAC was confirmed in WTPs. Appleman, et al. (2014) noted that long-chain PFCA and all PFSA were completely removed by GAC; however, short-chain PFCA in the same system had only 33%-91% removal. It was concluded that the limiting factor for GAC treatment was removal of short-chain PFBA and if short-chain PFAS are the target, GAC is not a preferred removal process. Likewise, Glover, et al. (2018) observed that longchain PFAS were removed well but breakthrough of short-chain PFAS was 48%-100% in the same system; also, targeted PFAS precursors were better removed than PFAA. This phenomenon was further confirmed by Eschauzier, et al. (2012) who observed that while PFOS and PFNA were removed, PFBA, PFPeA, PFHxA, PFOA, PFBS and PFHxS were not; shorter-chain PFAS dominated the post-GAC and finished drinking water. It was thought that short-chain PFBS was displaced by better-sorbing, longer-chain PFAA. Strikingly, breakthrough of short-chain PFBA was 10% after a

hydraulic loading of only 6,000 gallons per pound GAC. There is also a difference in sorption of branched vs. linear isomers in WTPs that was explained thermodynamically; linear PFAS isomers adsorb better than branched PFAS isomers (Eschauzier, et al., 2012).

Not only does PFAS speciation in the raw water affect GAC performance, but so do the presence of other constituents. The GAC surface must be positively charged to attract the negatively-charged PFAA. All PFAA are negatively charged relevant pH, so it is important that the raw water pH be less than the GAC pH point of zero charge. Additionally, other non-PFAS constituents in the influent water such as NOM and organic micropollutants may have a higher affinity for GAC and displace adsorbed PFAS.

While acidity and oxygen content of the GAC do not appear to affect PFAS sorption, the internal structure will affect diffusion of PFAS and impact overall sorption capacity. GAC source affects sorption capacity; bituminous coal shows better removal than natural materials (McNamera, et al., 2018). In every case, however, bench-scale tests should be performed under conditions relevant to the operation of the WTP. Utilities should consult with scientists and engineers familiar with designing such systems and perform treatability studies before undertaking full-scale design.

One of the shortcomings of using GAC for removal of PFAS, especially when short-chain and carboxylated PFAS are the target, is that it does not destroy PFAS but transfers them to a more concentrated medium. This is not a sustainable practice and GAC is an interim solution until full-scale destruction methods become practical. Residual management of contaminated GAC should be an early consideration.

Another issue is the rapid breakthrough of shortchain PFAS and consideration should be paid to the frequency and cost of regenerating and replacing the GAC. An independent engineering lifecycle cost analysis is critical during design. Burgess & Niple's analysis of several PFAS-impacted GAC systems in the Mid-Ohio Valley concluded that GAC vessels

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treating ammonium PFOA (C8) required replacement on average every 3 to 6 months; note that C8 is a longchain PFAA. This can cost hundreds of thousands of dollars per year in O&M. Rahman, et al. (2014) also found that changeout may be required as frequently as a matter of weeks for short-chain PFAA.

To accomplish thermal GAC regeneration, atmospheric release of PFAS must also be addressed. Change-out and regeneration of GAC must be timed to avoid breakthrough which requires a well-designed sampling and analysis plan. Analysis can also be a significant cost for GAC systems. Commercial prices of PFAS analysis are currently hundreds of dollars per sample.

Most of these issues also apply to PAC. It has been suggested that PAC be an emergency-response solution while GAC may be applied long term (Rahman, et al., 2014). Christian Eschauzier stated, "the option of reducing of the emissions from certain point sources...would appear more efficient than to spend money for a more frequent exchange of GAC..." (Eschauzier, et al., 2012). PFAS source control should be a consideration of any holistic PFAS solution. In the Cape Fear River Basin in North Carolina source considerations reduced influent PFAS concentrations to a nearby WTP by 90% in one month, reducing the burden on plant treatment technologies (Hopkins, et al., 2018).

Ion Exchange (IX)

IX resins are similar to GAC: they are adsorptive materials used full-scale for the removal of contaminants. IX relies on the affinity of the contaminant for the exchange site which can be chemically regenerated by washing with a solvent. Since PFAA are anionic at relevant pH, removal by anion exchange resins is possible. In fact, both electrostatic and hydrophobic interactions drive the adsorption of PFAS to IX resins (Rahman, et al., 2014). Porosity, polymer matrix, and functional groups are important considerations when choosing a resin. All of these impact the affinity of PFAS for the resin and its ability to reach the exchange sites. Merino, et al. (2016) published an extensive review on PFAS treatment methods including removal by commercially-available IX resins (i.e., Amberlite, Dow Marathon, and US Filter). The author concluded that under the appropriate conditions, IX can be more effective than GAC for removal of PFAS, especially for shorter-chain PFAA. Because of the enhanced hydrophilicity of acrylic resins over styrenic resins, the former are more effective for PFAS removal. Additionally, macroporous resins exhibit better uptake than gel resins because of better access to surface sites (Rahman, et al., 2014).

Composition of the raw water (e.g., PFAS speciation, NOM, competing ions) must also be considered when selecting a resin. Bench-scale and pilot-scale testing is essential for proper resin selection. In theory, resins can be regenerated using a mixture of a small amount of salt (ca. 1% to remove electrostaticallybound PFAS) and a large amount of alcohol (ca. 70% to remove hydrophobically-bound PFAS), and regenerated resins generally perform better than regenerated GAC (Merino, et al., 2016). In practice however, long regeneration times and large volumes of regenerant are required to remove the longer-chain PFAA; therefore, most IX for PFAS treatment are used as single-pass without regeneration. Some authors claim that traditional regeneration methods are ineffective for some PFAA (Carter and Ferrell, 2010). Given the complementary removal of PFAS using GAC and IX, a treatment train approach with GAC upstream of IX would likely be an effective treatment method. This has been accomplished in a system treating contaminated groundwater to sub ng/L levels (Woodward and Pearce, 2018).

IX performance in full-scale WTPs surveyed by Appleman, et al. (2014) and Rahman, et al. (2014) showed mixed results, often not agreeing with benchscale results. This is likely because conditions in bench-scale tests did not match those of the full-scale system. Each individual treatment system must be independently tested at bench- and pilot-scale to confirm performance before design and construction of a full-scale system. Another reason for mixed

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performance is that the IX systems surveyed were designed for NOM and metals removal and not for PFAS removal. Rahman, et al. (2014) reported that an iron-impregnated strong-base, porous IX resin achieved appreciable removal of PFCA (54% - 76%) and high removal of PFSA (83% - 93%, including short-chain PFBS), but failed to remove short-chain PFCA. This is consistent with results from a WTP in Alabama (Dickenson, et al., 2012). Appleman, et al. (2014) surveyed two WTPs which included the same IX resin used in Rahman, et al. (2014) and another metal-impregnated IX resin. In one plant, PFSA were removed better than PFCA (likely an electrostatic effect), and long-chain PFAA were removed better than short-chain PFAA (likely a hydrophobic effect). In the second WTP surveyed, IX showed little to no removal of PFAS analyzed. This was thought to be because the system's operational conditions were not specifically designed for PFAS removal.

Reverse Osmosis (RO)

Osmosis is the diffusion of water through a semipermeable membrane from a less-concentrated solution to a more concentrated one. Contrarily, RO is the movement of water in the opposite direction by the application of pressure. While osmosis is a thermodynamically spontaneous process, RO requires energy to operate.

Two liquid streams are produced during RO: the permeate (clean water) produced as water preferentially passes through the membrane while solutes do not, and the rejectate (wastewater) which is produced on the pressure side of the membrane where solutes concentrate. Generally, 80% permeate to 20% rejectate volume recovery is considered acceptable for WTPs. Of the three conventional water treatment technologies for PFAS removal, RO provides the best performance. RO systems with small pore sizes (<0.1 nm) are capable of substantial or complete PFAS removal, including short-chain PFAA not removed well by GAC and IX. This contrasts the performance of larger-pore size membranes surveyed by Appleman, et al. (2014), Rahman, et al. (2014), and Glover, et al. (2018) who reported essentially no removal of PFAS. The primary removal mechanism is physical due to PFAS molecular volume (about 1 nm) being larger than the membrane pore size (<0.1 nm). However, PFAS may also adsorb to the membrane itself by hydrophobic and/or electrostatic interactions (Liu, et al., 2018). So, while pore size of the membrane is arguably the most important design parameter, composition of the membrane surface is also important. High capital and energy costs and disposal of the concentrated rejectate are limitations of RO.

Rahman, et al. (2014) reported that charge-neutral PFAS precursors and some short-chain PFAA may not be well removed. This was also observed in a pilot-scale RO system deployed at Peterson Air Force Base in Colorado (Liu, et al., 2018). Appleman, et al. (2014) reported on the performance of two direct reuse RO systems in California using polyamide Hydraunautics ESPA2 and Toray and Hydraunautics membranes. At 12 gallons per square foot per day and 80% - 85% recovery, all PFAS analyzed were removed to below the method detection limit (MDL). The author concluded that RO was the most effective form of treatment evaluated. Two of the four WTPs surveyed in Glover, et al. (2018) employed RO. For both systems, average removals of PFAS studied ranged from 67±8% to 96±6%; these numbers are conservative as both WTPs removed PFAS to below MDLs of 1.0 ng/L or less.

EMERGING WATER TREATMENT TECHNOLOGIES FOR PFAS REMOVAL

Emerging treatment technologies for PFAS are at the bench-scale or pilot-scale. Many of these technologies are being developed for groundwater remediation and show promise as being applicable in full-scale WTPs.

Advanced Oxidation Processes (AOPs)

Oxidation processes involve the reaction of an oxidant with a reductant causing the reductant to lose an electron, breaking chemical bonds and leading to transformation or mineralization. AOPs employ highly oxidative and non-selective species to attack the reductant. PFAS play the role of reductant, and fluorine being the most electronegative atom in the periodic table is not likely to be oxidized by conventional oxidation processes. In fact, hypochlorite, permanganate and persulfate at ambient conditions, and ferrate(VI) are ineffective in transforming PFAS (Bhakri, et al., 2012). Other forms of ferrate (i.e., ferrate[IV] and ferrate[V]) have preliminarily shown transformation of PFOS and PFOA in model systems (Yates, et al., 2014). One metaphor for the perfluoroalkyl moiety describes the carbon chain as a piece of reinforcing steel, susceptible

to oxidation (i.e., rust) unless it is coated (e.g., by epoxy). Here the fluorine (epoxy) protects the carbon backbone (rebar) from oxidation (rust).

Several reviews have been published examining the effectiveness of AOPs on PFAS (Niu, et al., 2016; Wang, et al., 2017; Xu, et al., 2017; Dombrowski, et al., 2018; Schaefer, et al., 2018; Trojanowicz, et al., 2018; Nzeribe, et al. 2018). Some technologies which may become full-scale solutions for PFAS include:

- Direct Photolysis Oxidation by direct UV light;
- Photocatalysis Oxidation by radical species formed at a metal surface under UV light;
- Catalyzed Hydrogen Peroxide (CHP) Oxidation by hydroxyl radicals formed by H2O2 interaction with iron(II);
- Activated Persulfate Oxidation by the persulfate radical formed when persulfate is "activated" by UV light, microwave, heat, base, iron, or hydrogen peroxide;
- Electrochemical Oxidation Direct oxidation or oxidation by radicals at an electrode surface;
- Sonolysis Oxidation by intense heat and pressure within very small bubbles generated by acoustic waves;

- Radiolysis Direct oxidation by ionizing radiation; and
- Plasmolysis Oxidation by energy-induced plasma or free radicals (see Figure 3).

Most of these methods require significant investigation, design, cost, and life-cycle analysis before they can be considered practical at full-scale WTPs.

Figure 3. Lab-scale Plasma Reactor for PFAS Destruction. Courtesy of Dr. Michelle Crimi (Clarkson University)



Advanced Reductive Processes (ARPs)

Like AOPs, ARPs expose an oxidant (PFAS) to highlyreactive reductive radicals transferring electrons to the oxidant, thereby breaking chemical bonds. Reductive radicals which have been shown effective for PFAS destruction include solvated electrons, reductive hydrogen atom, reductive sulfate radicals, and reductive iodide radicals (Merino, et al., 2014; Nzeribe, et al., 2018). Reductive radicals for PFAS degradation have been formed by dithionite, aqueous iodide, and ferrocyanide in combination with UV light, laser flash photolysis, ultrasound, microwave, and electron beam (E-Beam). The use of some chemicals for production of the reductive radicals are not appropriate for drinking water because of their cost and toxicity. These methods also require significant investigation, design, cost, and life-cycle analysis before they can be considered practical at full-scale WTPs.

Next-Generation Adsorbents

Many researchers are attempting to improve on the success of GAC and IX by producing new and modifying existing absorbents. Several reviews have been published with extensive data on batch adsorption tests and column studies, to determine practicality of application at full-scale WTPs (Du, et al., 2014; Omo-Okoro, et al., 2018; Oyetade, O.A., et al., 2018). Some next generation adsorbents that have shown promise include:

- Tailored GAC and IX resins;
- Carbon nanotubes;
- Graphene;
- · Organically-modified silica;
- Organo-clays;
- Molecularly-imprinted polymers;
- · Cationic/anionic surfactants;
- Black carbon;
- Magnetic mesoporous carbon nitride;
- Polymeric absorbents;
- Mesoporous molecular sieves;
- Organic frameworks;
- Permanently-confined micelle arrays; and
- Electrocoagulation and removal on metal hydroxide flocs.

Most of these adsorbents are too expensive to be applicable to full-scale WTPs. However, research into low-cost methods may one day make these practical at full-scale WTPs.

Biodegradation

Biotransformation of PFAS precursors by bacteria, fungi and isolated enzymes has been observed under aerobic conditions in wastewater and in the environment (Butt, et al., 2014). These transformations convert PFAS precursors to PFAA but do not destroy the perfluoroalkyl moiety. This was also suggested to be occurring within full-scale BAC reactors reviewed by Appleman, et al. (2014) and Glover, et al. (2018). There is shortening of the perfluoroalkyl moiety during biotransformation of PFAS precursors, however, a stoichiometric mass of PFAA is formed (Ross, et al., 2018).

PFAA, however, have not been shown to mineralize under aerobic conditions. Anaerobic degradation of PFOS and PFOA in a lab-scale wastewater bioreactor has been reported (Meesters and Schroeder, 2004); however, it is unclear if the mass reductions observed were due to mineralization or sorption. Complete biological mineralization of any of the thousands of PFAS has not been reported (Ross, et al., 2018). To date, no demonstration of PFAA degradation under conditions relevant to drinking water applications have been published. That is not to say that this is impossible. If possible, microbial metabolism of PFAA will probably be like reductive dechlorination and will fit within the umbrella of reductive dehalogenation. At one time it was thought that biodegradation of polychlorinated biphenyls (PCBs) was not possible. However, the groundbreaking work of Dr. Lisa Alvarez-Cohen and others led to an entire industry of bioremediation for PCBs.

SUMMARY AND CONCLUSIONS

This article presents a brief survey of effective and ineffective full-scale water treatment technologies for their removal. Some emerging technologies have also been mentioned that one day may be practical at full-scale WTPs. The key points of this article include:

- Most conventional drinking water treatment technologies are ineffective in removing PFAS from contaminated raw water sources except for granular and powder activated carbons (GAC and PAC), ion exchange (IX), and reverse osmosis (RO);
- Use of GAC, IX and RO may not remove all target PFAS, especially short-chain, and carboxylated PFAA;
- Any system considered for the removal of PFAS in drinking water requires independent bench- and pilot-scale testing under relevant conditions before undertaking fullscale design;
- While no technologies exist for mineralization within full-scale water treatment plants (WTPs), several emerging technologies show promise and further research may make practical their use at fullscale WTPs;
- PFAS source removal and reduction to raw water sources should be part of any plan to eliminate human exposure via drinking water.

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REFERENCES

Reference list and select references can be obtained by contacting the author at bryan. yates@burgessniple.com or 614-459-2050.