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Date: February 19, 2025

To: Rep. Amy Sheldon, Chair
House Committee on Environment

cc: Rep. Larry Labor, Vice Chair
Rep. Larry Satcowitz, Ranking Member
Rep. Sarah "Sarita" Austin, Clerk
Rep. John L. Bartholomew
Rep. Ela Chapin
Rep. Kate Logan
Rep. Kristi Morris
Rep. Rob North
Rep. Christopher "Chris" Pritchard
Rep. Michael "Mike" Tagliavia
Anika Adams, Committee Assistant

From: Martin Wolf, Principal
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and

Advisor, Safer Chemicals and Circular Economy
The American Sustainable Business Network
Washington, DC 20002

RE: Testimony on H. 238 - An act relating to the phaseout of consumer products containing added perfluoroalkyl and polyfluoroalkyl substances

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Dear Rep. Sheldon and Committee Members:

Thank you for this opportunity to testify on H. 238 - an act relating to the phaseout of consumer products containing added perfluoroalkyl and polyfluoroalkyl substances (PFAS).

For the record, my name is Martin Wolf, Principal, Wolf Sustainability. I am a chemist with over 50 years of industry experience studying the occurrence and fate of chemicals in the environment and designing more sustainable consumer products.

I also represent the American Sustainable Business Network, a multi-issue membership organization advocating on behalf of businesses, business associations, and the investor community, whose members collectively represent over 250,000 businesses.

Summary of Recommendations

It is recommended that:

a) the definition of PFAS in H.238 be retained,
b) a de minimis level of PFAS in consumer products be established, and
c) the manufacture, sale, or distribution for sale of consumer products containing PFAS ultimately be prohibited.

Human and Environmental Harms of PFAS

Perfluoroalkyl and polyfluoroalkyl substances (PFAS) are substances containing fluorine atoms covalently bonded to carbon atoms. PFAS are known to be harmful to human health and have been associated with a variety of health effects, including altered immune and thyroid function, liver disease, lipid and insulin dysregulation, kidney disease, adverse reproductive and developmental outcomes, and cancer.[1] Concordance with experimental animal data exists for many of these effects.[2]

PFAS have been detected in rain, snow, groundwater, tap water, lakes, rivers, soils, and sediments and are ubiquitous in the environment.[3] PFAS degradation products are freely mobile in water, soil, and air, and can be extremely resistant to breakdown.[4] Given their potential to cause harm, ubiquity and resistance to breakdown, it is vitally important that PFAS uses and emissions must be rapidly restricted.[5]

Economic Costs of PFAS

For most consumer applications, the economic impacts of remediating harm to human health and harm to the environment from PFAS far exceed the economic value of transitioning to less hazardous alternatives. The benefit to PFAS manufacturers has been estimated as $2billion per year.[6] Health-related costs for the United States are estimated to be $37−59 billion annually, not including indirect social costs such as lost wages; lost years of life; reduced quality of life;

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[1] Fenton SE, Ducatman A, Boobis A, DeWitt JC, Lau C, Ng C, Smith JS, Roberts SM. Per- and Polyfluoroalkyl Substance Toxicity and Human Health Review: Current State of Knowledge and Strategies for Informing Future Research. Environ Toxicol Chem. 2021 Mar;40(3):606-630. doi: 10.1002/etc.4890. Epub 2020 Dec 7. PMID: 33017053; PMCID: PMC7906952.
[2] Ibid.
[3] Abunada, Z.; Alazaiza, M.Y.D.; Bashir, M.J.K. An Overview of Per- and Polyfluoroalkyl Substances (PFAS) in the Environment: Source, Fate, Risk and Regulations. Water 2020, 12, 3590. https://doi.org/10.3390/w12123590
[4] Ibid
[5] Environ. Sci. Technol. 2022, 56, 16, 11172–11179
[6] Alissa Cordner, Gretta Goldenman, Linda S. Birnbaum, Phil Brown, Mark F. Miller, Rosie Mueller, Sharyle Patton, Derrick H. Salvatore, and Leonardo Trasande, *The True Cost of PFAS and the Benefits of Acting Now*, Environmental Science & Technology 2021 55 (14), 9630-9633. DOI: 10.1021/acs.est.1c03565

increased stress, anxiety, and depression; and subsequent impacts on families and communities.[7] Additionally, the Minnesota Pollution Control Agency estimates the cost to remove just one pound of PFAS from wastewater, biosolids, municipal solid waste, or landfill leachate to be between $0.4 million and $39 million depending on the matrix and size of facility.[8] Thus, there is not an economic case for continued use of PFAS and it is recommended that their use be phased out rapidly to limit future remediation costs.

Definitions

The exact definition of PFAS has varied.[9] The definition of PFAS in H.238, "…a class of fluorinated organic chemicals containing at least one fully fluorinated carbon atom" is preferred to the definition proposed in the Agency of Natural Resources report dated November, 2024, taken from the Code of Federal Regulations, "PFAS means any chemical substance or mixture containing a chemical substance that structurally contains at least one of the following three sub-structures:

(1) $R-(CF_2)-CF(R')R''$, where both the CF_2 and CF moieties are saturated carbons.

(2) $R-CF_2OCF_2-R'$, where R and R' can either be F, O, or saturated carbons.

(3) $CF_3C(CF_3)R'R''$, where R' and R" can either be F or saturated carbons."[10]

This definition has four major limitations: (1) omission of substances that have functional groups on both ends of the fully fluorinated carbon moiety (e.g., perfluoroalkyldicarboxylic acids); (2) inconsistencies in dealing with homologues that are fully fluorinated aliphatic cyclic compounds with or without a fully fluorinated alkyl side chain; (3) omission of substances with aromatic ring(s) in the nonfluorinated functional group(s) that can be cleaved in the environment and biota; and (4) use of the ambiguous term "highly fluorinated".[11] Many substances in commerce, known to harm human health and the environment, are omitted by ANR's proposed definition.[12]

De Minimis Level

Companies are appropriately concerned that their products may be held to a higher standard than municipal, ground, or surface water used to manufacture their products. According to the Agency for Toxic Substances and Disease Registry (ATSDR) ingestion of food and water is a main

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[7] Ibid.

[8] Minnesota Pollution Control Agency, Evaluation of Current Alternatives and Estimated Cost Curves for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water, 2023.

[9] Zhanyun Wang, Andreas M. Buser, Ian T. Cousins, Silvia Demattio, Wiebke Drost, Olof Johansson, Koichi Ohno, Grace Patlewicz, Ann M. Richard, Glen W. Walker, Graham S. White, and Eeva Leinala*, A New OECD Definition for Per- and Polyfluoroalkyl Substances,* Environmental Science & Technology 2021 55 (23), 15575-15578

[10] 40 CFR 705.3 "Per- and polyfluoroalkyl substances or PFAS"

[11] Zhanyun Wang, et al., ibid.

[12] Ibid.

route of PFAS exposure.[13] Therefore it is recommended that a de minimis thresholds be established by the Secretary for PFAS in a product or a product component, and that the threshold be higher than the maximum contaminant level (MCL) of PFAS set for municipal drinking water,[14] ground water, or surface water.[15]

Conclusion

PFAS represent an immediate threat to the health of Vermont's citizens and our environment. H.238 takes important steps to limiting this threat by using a scientifically valid, inclusive definition of PFAS and by phasing out PFAS in a wide range of consumer products. I commend this Committee, ANR, and the other State agencies involved in developing this legislation.

Thank you for your attention to, and consideration of, these comments.

Respectfully submitted,

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and

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[13] Human Exposure: PFAS Information for Clinicians - 2024 | PFAS and Your Health | ATSDR. Downloaded 12 February 2025

[14] US EPA, PFAS National Primary Drinking Water Regulation, 2024.

[15] US EPA, DRAFT Human Health Ambient Water Quality Criteria: Perfluorooctanoic Acid (PFOA) and Related Salts 2024.

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Per- and Polyfluoroalkyl Substance Toxicity and Human Health Review: Current State of Knowledge and Strategies for Informing Future Research

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Abstract

Reports of environmental and human health impacts of per- and polyfluoroalkyl substances (PFAS) have greatly increased in the peer-reviewed literature. The goals of the present review are to assess the state of the science regarding toxicological effects of PFAS and to develop strategies for advancing knowledge on the health effects of this large family of chemicals. Currently, much of the toxicity data available for PFAS are for a handful of chemicals, primarily legacy PFAS such as perfluorooctanoic acid and perfluorooctane sulfonate. Epidemiological studies have revealed associations between exposure to specific PFAS and a variety of health effects, including altered immune and thyroid function, liver disease, lipid and insulin dysregulation, kidney disease,

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adverse reproductive and developmental outcomes, and cancer. Concordance with experimental animal data exists for many of these effects. However, information on modes of action and adverse outcome pathways must be expanded, and profound differences in PFAS toxicokinetic properties must be considered in understanding differences in responses between the sexes and among species and life stages. With many health effects noted for a relatively few example compounds and hundreds of other PFAS in commerce lacking toxicity data, more contemporary and high-throughput approaches such as read-across, molecular dynamics, and protein modeling are proposed to accelerate the development of toxicity information on emerging and legacy PFAS, individually and as mixtures. In addition, an appropriate degree of precaution, given what is already known from the PFAS examples noted, may be needed to protect human health.

Keywords

Per- and polyfluoroalkyl substances; Perfluorooctane sulfonate; Perfluorooctanoic acid; Persistent compounds; Contaminants of emerging concern

INTRODUCTION

Per- and polyfluoroalkyl substances (PFAS) are ubiquitous in environmental media because of their prolific use in a variety of industrial and consumer products and processes (Jian et al. 2018; Sunderland et al. 2019). Widespread human exposure to PFAS in water, food, and air coupled with the lengthy environmental persistence and biological half-lives of some PFAS have led to measurable PFAS in the blood of nearly the entire population in developed countries, with health effects reported globally (Kato et al. 2011; Khalil et al. 2016; Stubleski et al. 2016; Jian et al. 2018). Information needed to evaluate the potential risk of harm from PFAS includes the types of adverse health effects that might occur at environmentally relevant exposures, especially in sensitive life stages. Information is also needed regarding the mode(s) of action for PFAS toxicity, PFAS toxicokinetics in both humans and laboratory animal models, and dose-response relationships. Risk estimates can be used to inform public health exposure limits that will determine the need for exposure mitigation and environmental cleanup.

There are several challenges in obtaining the information needed to assess human health risk from the large number of PFAS with a wide range of structures and chemical properties (Buck et al. 2011; Wang Z et al. 2017; Organisation for Economic Co-operation Development 2018). Data on the identity, composition, and quantity of PFAS used in products and processes are often treated as confidential business information, hampering efforts to estimate exposure sources and routes. The Organisation for Economic Co-operation and Development's (OECD's) chemical inventory reports over 4000 substances that contain at least one perfluoroalkyl (–CnF2n–) moiety (Organisation for Economic Co-operation Development 2018), and the US Environmental Protection Agency (USEPA) has a curated list of over 8000 PFAS included, based on structure (US Environmental Protection Agency 2018) from the CompTox Chemicals Dashboard (Williams et al. 2017). The USEPA estimates that more than 600 PFAS are currently in commercial use (US Environmental Protection Agency 2019). Experimental studies of PFAS have been limited by funding and the availability of analytical standards, confounded by the prevalence of background contamination in laboratory materials, and challenged by physicochemical properties such as high surface activity that can interfere with and complicate measurements. Consequently, sufficient information to conduct quantitative risk assessment is currently available for only a relative few PFAS (Post 2020). Further, although typical human exposures involve various combinations of PFAS (Centers for Disease Control and Prevention 2017), only a few efforts address interactions of PFAS mixtures; and a well-founded, scientific basis on which to evaluate their combined toxic potential does not yet exist (Carr et al. 2013; Wolf et al. 2014; Zhou et al. 2017; Hoover et al. 2019; US Environmental Protection Agency 2020).

The Society of Environmental Toxicology and Chemistry (SETAC) North America held the focused topic meeting and workshop “Environmental Risk Assessment of PFAS” on 12 to 15 August 2019, covering a wide range of topics related to the characterization of health risks posed by PFAS. The overarching purpose of the meeting was to begin a scientific discussion on how best to approach studying, grouping, and regulating the large number of PFAS to which people and other species are potentially exposed (for charge questions and other details, see Johnson et al. 2020). We refer to these PFAS as “legacy” (those perfluoroalkyl acids for which there are accumulating health data but that may be phased out or decreased in use) and “emerging” (those which are being used as replacements, often with minimal health effects data). The objectives of the Human Health Toxicity section were to provide an assessment of the state of the science in understanding toxicological effects of PFAS and to explore and discuss strategies for advancing knowledge on the toxicity of individual and groups of PFAS.

CURRENT KNOWLEDGE OF PFAS TOXICITY IN HUMANS

Like other chemicals, PFAS are potentially capable of producing a wide range of adverse health effects depending on the circumstances of exposure (magnitude, duration, and route of exposures, etc.) and factors associated with the individuals exposed (e.g., age, sex, ethnicity, health status, and genetic predisposition). Aspects to consider when establishing the health effects of greatest concern are 1) effects for which evidence is the strongest (strength of evidence can come from consistency of effect across studies, strength of effect associations in epidemiological studies, and species concordance, as examples), and 2) effects for which potential impact is greatest (factors contributing to impact can include severity of effect, functional impairment, persistence, and specific age groups that are susceptible, as examples). Brief summaries of candidate PFAS health effects from human and experimental reports are provided in this section (Figure 1).

Immune function

Epidemiological studies have explored relationships between PFAS exposure and laboratory biomarkers of immunomodulation, such as vaccine responses. A doubling of perfluorooctane sulfonate (PFOS) in maternal serum was associated with a 39% ($p < 0.001$) reduction in diphtheria antibody concentration in children (age 5 yr), with increased odds of falling below clinically protective values against diphtheria and tetanus at age 7 yr. The authors noted that a “2-fold greater concentration of major PFCs [perfluorinated compounds] in child serum was associated with a difference of −49% (95% CI, −67% to −23%) in the overall antibody concentration" (Grandjean et al. 2012). Decreased immunological response persisted at age 13 yr (Grandjean et al. 2017). Adverse associations were also noted for responses to rubella, mumps, and *Hemophilus* influenza vaccinations in children and to vaccinations in adults (Granum et al. 2013; Looker et al. 2014; Stein et al. 2016; Abraham et al. 2020). In a single study, modest down-regulation of C-reactive protein response, a marker of human systemic inflammation, was also reported to be associated with perfluorooctanoic acid (PFOA) blood levels (Genser et al. 2015).

Disease outcomes linked with immunosuppression such as clinician-recorded diagnoses of childhood infections have also been associated with prenatal exposures to PFOS and perfluorohexane sulfonate (PFHxS) (Goudarzi et al. 2017). A pregnancy cohort study prospectively detected increased risk of airway and throat infections and diarrhea in children through age 10 yr, correlated with cord-blood PFAS measurements (Impinen et al. 2018, 2019). A recent review concluded that exposure to PFAS in infancy and childhood resulted in an immunosuppressive effect characterized by an increased incidence of atopic dermatitis and lower respiratory tract infections (Kvalem et al. 2020). Some of the immunological effects were sex-specific, but the authors cautioned that there were inconsistencies across studies (Kvalem et al. 2020). Overall, available data provide strong evidence that PFAS exposure can suppress the human immune response.

Population studies of immune hyperreactive diseases have resulted in mixed findings. Studies on childhood allergy and asthma outcomes have shown no association with PFAS (Impinen et al. 2018, 2019), whereas others have found substantial effects, including provocative evidence that subgroups of individuals not adequately immunized may be at an increased risk for disease a priori (Qin et al. 2017; Timmermann et al. 2017a). For example, a case-control study of Taiwanese children compared the first and fourth quartiles of serum measurements for 11 PFAS with asthma and other immune markers and reported confidence intervals well above 1.0 for PFOA and others (Qin et al. 2017). However, review articles concerning PFAS and childhood allergy and asthma offer nuanced, age- and sex-specific interpretations and advise against firm conclusions (Kvalem et al. 2020).

Chronic autoimmune outcomes, including thyroid disease (see section Thyroid function) and inflammatory bowel disease (IBD), have also been considered. A study in contaminated communities ($n = 32\,254$) detected an association between both prevalence and incidence of ulcerative colitis (UC) and PFOA exposure (linear trend $p = 0.0001$ [Steenland et al. 2013]). A worker study ($n = 3713$) found a higher prevalence ($p = 0.01$) and incidence ($p < 0.05$) of UC with increasing log PFOA serum concentrations (Steenland et al. 2015). A case-control study of children and young adults from a background exposure community in Atlanta, Georgia, USA, also found higher serum PFOA levels in patients with UC (Steenland et al. 2018b). In contrast to PFOA-related associations in US populations, a study of a contaminated community in Sweden ($n = 63\,074$) did not show a consistent association of IBD with any PFAS exposure (Xu et al. 2020b).

Recent, thorough reviews (National Toxicology Program 2016; DeWitt et al. 2019; Pachkowski et al. 2019) emphasize some key concepts: 1) there is concordance between animal studies and human epidemiological observations that PFAS modify the immune response, and 2) there are noted complexities in assuming dose-response continuums, including possible differences in life-stage vulnerability. Authors of these reviews note uncertainty about which outcome will be of most importance but agree that immunotoxicity should be included among sensitive human PFAS toxicity endpoints.

Thyroid function

The C8 Science Panelists concluded that there is a “probable link” of PFOA exposure to thyroid disease, with sex-specific outcomes in women (for hyperthyroid disease) versus men (hypothyroid disease) (C8 Science Panel 2012). Subsequent reviews drew attention to hypothyroid outcomes in women and children and to the possibility that populations with a priori circulating antithyroid peroxidase antibodies may be at additional risk (Coperchini et al. 2017). A broad childhood disease review noted “some evidence” that PFAS cause childhood hypothyroidism and characterized the number of studies as “limited” for childhood disease conclusions (Rappazzo et al. 2017). A meta-analysis of 12 child and adult studies that excluded populations with higher exposures noted that PFAS exposure is negatively associated with serum total thyroxine levels and that “PFAS could induce thyroid dysfunction and disease” (Lee and Choi 2017).

Human thyroid disease is mostly the result of an autoimmune response and is 5 to 10 times more prevalent in women than men (Tadic et al. 2018). Concerning PFAS and clinically diagnosed outcomes, women in the highest quartile of PFOA exposure (>5.7 ng/mL) reported clinical hypothyroid disease (odds ratio 2.2, 95% confidence interval [CI] 1.4–3.7) over 3 cycles of National Health and Nutrition Examination Survey (NHANES) data (1999–2006, $n = 3974$ adults), with similar findings in men (Melzer et al. 2010). The C8 Science Panel studies (median serum PFOA 26.1 ng/mL) found thyroid disease hazard ratios of 1.00, 1.24, 1.27, 1.36, and 1.37 across cumulative exposure quintiles in women (log-linear trend $p = 0.03$ [Winquist and Steenland 2014b]), with parallel hypothyroid findings in children aged 1 to 17 yr (Lopez-Espinosa et al. 2012). The Ronneby, Sweden, population experienced excess risk of thyroid disease in a discrete time period (1984–2005) among women (hazard ratio 1.29, 95% CI 1.05–1.57) that did not persist over time despite higher cumulative PFAS exposure (Andersson et al. 2019). The authors did not link exposure to hypothyroid outcome, noting a nonmonotonic dose-response relationship (Andersson et al. 2019).

Human population studies augment experimental data that PFAS interact with thyroid hormone binding proteins (Berg et al. 2015; Ren et al. 2016; Zhang J et al. 2016), one of several mechanisms by which PFAS can perturb feedback relationships between free thyroid hormone and the hypothalamic-pituitary-thyroid axis. Exposures to PFAS also interfere with thyroid peroxidase (TPO) enzyme activity in vitro (Song et al. 2012). Several PFAS studies have pursued this putative mechanism, finding that maternal and neonatal thyroid hormone outcomes were more readily detected in those with a priori abnormally high circulating anti-TPO antibodies (Webster et al. 2014, 2016). One case-control study investigated congenital hypothyroidism, a rare condition. Serum concentrations of PFOA (5.40 vs 2.12 ng/mL; $p < 0.01$), perfluorononanoic acid (PFNA; 1.93 vs 0.63 ng/mL; $p < 0.001$), perfluorodecanoic acid (PFDA; 0.52 vs 0.30 ng/mL; $p < 0.005$), and perfluoroundecanoic acid (0.98 vs 0.44

ng/mL; $p < 0.005$) were higher in the diagnosed newborns; and levels of several PFAS, including PFOA and PFHxS, were correlated with thyroid autoantibodies (Kim et al. 2016).

Thyroid disease is not the only concern. Clinicians are concerned about subclinically elevated thyroid-stimulating hormone (TSH) in early pregnancy because it may be associated with several possible adverse maternal and fetal outcomes (Forhead and Fowden 2014). This general concern has prompted numerous PFAS-exposure evaluations of corresponding TSH in maternal serum, cord blood, and newborns. A review of maternal and child biomarkers with PFAS exposure noted that higher TSH has been reported in 4 second-trimester studies (Ballesteros et al. 2017), but there are also conflicting findings. Studies measuring PFAS in the first trimester have also found associations between PFAS exposure and altered TSH levels in newborns, including nonmonotonic patterns of dose response that mirror the marked alterations of thyroid hormone levels during pregnancy (Inoue et al. 2019).

From the available studies, PFAS definitively alter human thyroid hormones and potentially contribute to thyroid auto-immunity but do not so far appear to be a cause of thyroid cancer (Barry et al. 2013; Vieira et al. 2013). Also, thyroid cancer is usually survived; thus, morbidity rather than mortality studies are useful.

Liver disease and cancer

The liver is a primary target organ for long-chain PFAS storage, and accompanying experimental evidence of toxicity includes hepatocyte fat infiltration, specific P450 (CYP) pathway induction, apoptosis, hepatocellular adenomas and carcinomas, and disrupted fatty acid trafficking that can be peroxisome proliferator-activated receptor alpha (PPARα)-dependent or -independent and present across species (Maestri et al. 2006; Cui et al. 2009; Wan et al. 2012; Huang et al. 2013; Perez et al. 2013; Filgo et al. 2015; Xu et al. 2016, 2020a; Yao et al. 2016; Zhang L et al. 2016b; Hui et al. 2017; Li et al. 2017a; Guillette et al. 2020; National Toxicology Program 2020a).

Population studies demonstrate significant associations of long-chain PFAS (>6 fluorinated carbons) exposure to higher liver enzymes, such as alanine aminotransferase in adults and adolescents (Sakr et al. 2007a; Gallo et al. 2012; Yamaguchi et al. 2013; Gleason et al. 2015; Attanasio 2019; Nian et al. 2019), including in longitudinal studies (Sakr et al. 2007b; Darrow et al. 2016). Following low-dose exposures, these associations may be more evident in obese participants (Lin et al. 2010; Gallo et al. 2012; Jain and Ducatman 2019e).

Based on experimental data (Martin et al. 2007; Wan et al. 2012; Wang et al. 2013; Das et al. 2017), nonalcoholic fatty liver disease (NAFLD) has been investigated as a clinical outcome of PFAS exposure mediating consistent population PFAS-altered liver enzyme findings. Studies with NAFLD cytokeratin C18 biomarkers have provided supportive evidence for PFAS inducing steatosis (Bassler et al. 2019). Metabolomic studies have been directed at potentially explanatory human glycerophosphocholine and fatty acid profiles (Kingsley et al. 2019; Salihovic et al. 2019; Wahlang et al. 2019). Processes which favor steatosis promote advanced liver disease including liver cancer in humans (Massoud and Charlton 2018; National Toxicology Program 2020a). Associations of PFAS with advanced human liver disease and liver cancer are technically hard to study for reasons including (and not limited to) lethality, selection of comparison populations, and alterations of excretion mechanics associated with disease states. In a clinic-based study, mostly obese (85%) children aged 7 to 19 yr with biopsy-proven NAFLD had more advanced disease associated with PFOS and PFHxS exposure as well as associations with lipid and amino acid pathways linked to NAFLD pathogenesis (Jin et al. 2020). However, an adult study reported that serum PFHxS was inversely associated with hepatic lobular inflammation in morbidly obese bariatric surgery patients (Rantakokko et al. 2015). A study of heavily exposed workers ($n = 462$, geometric mean serum PFOA of 4048 ng/mL) detected significantly increased incident mortality for cirrhosis (relative risk = 3.87, 95% CI 1.18–12.7) and liver cancer (relative risk = 6.69, 95% CI 1.71–26.2) compared to a regional population (Girardi and Merler 2019), whereas no PFAS association to cancer or advanced liver disease was reported in a 3M worker cohort or in the C8 Health study population (Lundin et al. 2009; Barry et al. 2013; Vieira et al. 2013).

Emerging animal toxicology and histology and human population data provide mechanistic clues that PFAS disrupt hepatic metabolism, leading to increased bile acid reuptake and lipid accumulation in liver (Salihovic et al. 2020; Schlezinger et al. 2020). A review of NAFLD and toxicant exposure concluded that PFAS are associated with early steatosis (“fatty liver”), the preclinical stage of NAFLD (Armstrong and Guo 2019).

Lipid and insulin dysregulation

Cross-sectional and longitudinal investigations indicate that PFAS increase serum total and low-density lipoprotein cholesterol in adults and children (Steenland et al. 2009; Frisbee et al. 2010; Nelson et al. 2010; Eriksen et al. 2013; Fisher et al. 2013; Fitz-Simon et al. 2013; Geiger et al. 2013; Fu et al. 2014; Starling et al. 2014; Winquist and Steenland 2014a; Skuladottir et al. 2015; Zeng et al. 2015; Koshy et al. 2017; Convertino et al. 2018; He et al. 2018; Seo et al. 2018; Dong et al. 2019; Lin et al. 2019; Li et al. 2020; Liu G et al. 2020), including clinically defined high cholesterol (Steenland et al. 2009; Winquist and Steenland 2014a; Lin et al. 2019). Studies of large populations, featuring wide exposure ranges, demonstrate that serum lipids rapidly increase beginning at background (1–10 ng/mL) serum concentration and then are followed by attenuating (“plateaued”) cholesterol measurements as (log-transformed) exposures to long-chain PFAS increase (Steenland et al. 2009; Frisbee et al. 2010; Li et al. 2020). These findings suggest partially saturable mechanisms; thus, the cholesterol dose response at pharmacologic or acutely toxic doses should be viewed with caution; associations can be missed or may be misleading when an environmental range of exposure is absent. At background exposure levels, residual associations may be more detectable in obese participants (Timmermann et al. 2014; Jain and Ducatman 2019d), a finding congruent with experimental PFAS outcomes in rodents fed “Western” or high-fat diets (Tan et al. 2013; Quist et al. 2015; Rebholz et al. 2016). Human gene expression pathways provide support for an interaction of obesity and PFAS exposures and suggest possible sex differences (Fletcher et al. 2013). A pharmacokinetic model predicts that approximately half of the PFOS-exposed population would experience a >20% rise in serum cholesterol (Chou and Lin 2020). Risk-assessment implications for low-PFAS dose increases in cholesterol have been noted (New Jersey Drinking Water Quality Institute Health Effects

Subcommittee 2017; Li et al. 2020), and a review of population and toxicity data concluded that dyslipidemia is the strongest metabolic outcome of PFAS exposure (Sunderland et al. 2019).

Human PFAS lipid findings may be related to experimental findings of induced adipogenesis, impaired bile acid metabolism/synthesis, strongly decreased CYP7A1 enzyme activity, altered fatty acid transport, and intracellular lipid accumulation with steatosis, including in PPAR-α-null or PPAR-α-humanized animals (Guruge et al. 2006; Lau et al. 2007; Bijland et al. 2011; Bjork et al. 2011; Wang et al. 2014; Filgo et al. 2015; Das et al. 2017; Salihovic et al. 2019; Zhang et al. 2019; Behr et al. 2020a; Liu S et al. 2020b; Schlezinger et al. 2020). Independent of PFAS exposure, similar alterations in metabolic pathways have been related to disrupted fatty acid beta-oxidation and increased free cholesterol in toxicology studies (Perla et al. 2017).

Cross-sectional studies of diabetes outcomes can be misleading for reasons discussed in the renal section (see section Kidney disease, uric acid, and kidney cancer). Emerging longitudinal and diabetes clinical trial data indicate that PFAS may increase human insulin resistance, associated with dysregulated lipogenesis activity (Alderete et al. 2019; Lin et al. 2019). Longitudinal studies of clinically diagnosed diabetes patients have sometimes associated PFAS exposures with diabetes (Sun et al. 2018) or with small changes in glycemic markers (Cardenas et al. 2017); however, diabetes associations to date are not consistent (Karnes et al. 2014; Cardenas et al. 2017; Donat-Vargas et al. 2019). Future studies should consider whether PFAS may instigate autoimmune diabetic outcomes in humans, as shown in experimental studies (Bodin et al. 2016). Experimental data reveal that PFAS activate G protein-coupled receptor 40, a free fatty acid-regulated membrane receptor on islet ß cells, stimulating insulin secretion (Qin et al. 2020; Zhang L et al. 2020).

Kidney disease, uric acid, and kidney cancer

Extended human half-lives of long-chain PFAS are attributed to active renal tubular reabsorption. Of concern, legacy PFAS such as PFOA and PFOS are concentrated in renal tissues, and histopathologic, molecular, oxidative stress, and epigenetic studies provide evidence of potential nephrotoxicity (Wen et al. 2016; Stanifer et al. 2018; Sakuma et al. 2019; Rashid et al. 2020). In addition, the strong influence of kidney reabsorption on the extended half-lives of long-chain PFAS is consistent with both human protein binding and experimental PFAS excretion data.

Human studies have associated legacy PFAS exposure to diminished glomerular filtration and/or defined chronic kidney disease in adults and children (Shankar et al. 2011; Watkins et al. 2013; Kataria et al. 2015; Blake et al. 2018). However, this outcome may be due to reverse causation (Watkins et al. 2013; Dhingra et al. 2017). Some reviews of the available epidemiologic and toxicologic evidence suggest causative links between PFAS and diminished kidney function and chronic kidney disease (Stanifer et al. 2018; Ferrari et al. 2019); these authors also note several knowledge gaps and uncertainty about which proposed mechanisms of action are most important. A propensity score approach to NHANES data (Jain and Ducatman 2019c; Zhao et al. 2020) and a study with repeated

PFAS and health measures over an 18-yr period (Blake et al. 2018) recently concluded that PFAS exposure likely causes diminished renal glomerular filtration.

Uric acid, a biomarker of increased risk for renal disease (Obermayr et al. 2008), is also consistently associated with PFAS exposure in adults and children (Steenland et al. 2010; Geiger et al. 2013; Gleason et al. 2015; Kataria et al. 2015; Qin et al. 2016; Zeng et al. 2019), including a visible dose-response curve that begins at or near historic background levels in human populations (Steenland et al. 2010; Zeng et al. 2019). Serum PFAS concentrations exhibit an inverted U-shaped pattern related to glomerular filtration, initially exhibiting a modest accumulation as glomerular filtration begins to decrease and then decreasing in advancing renal disease, likely due to failure of normal strong reabsorption mechanisms in moderate to severe kidney disease (Jain and Ducatman 2019c). This finding is more dramatic across stages of glomerular filtration when there is also albuminuria (Jain and Ducatman 2019b). Studies suggest that the association of PFAS to uric acid is not due to reverse causation and is underestimated because the failing kidney excretes long-chain PFAS but retains uric acid. An implication is that population outcomes that occur in the presence of either albuminuria or moderate to severe renal disease such as hypertension (Jain 2020) increasing presence of and uric acid (a biomarker of renal disease; Jain and Ducatman 2019a; Zeng et al. 2019) can be underestimated in cross-sectional studies; in other words, the link between these health outcomes and PFAS exposure is obscured in these studies because of enhanced PFAS excretion patterns in the presence of either albuminuria or moderate to severe kidney disease. Furthermore, the strong influence of renal reabsorption on the long half-lives of long chain PFAS is consistent with both human protein binding of PFAS and experimental PFAS excretion rates in high-dose rodent studies (Cheng and Ng 2017).

Kidney cancer diagnoses have been increasing since 1975, a finding that is partially independent of improved detection, with 5-yr cancer-specific survival of approximately 80% (Gandaglia et al. 2014). The C8 Health studies noted longitudinal ($n = 32\ 254$) increases of kidney cancer (hazard ratio = 1.10, 95% CI 0.98–1.24) and kidney cancer mortality (Steenland and Woskie 2012; Barry et al. 2013; Vieira et al. 2013). A review of 6 published studies found long-chain PFAS exposure associated with kidney cancer or kidney cancer mortality, with risks ranging from 1.07 to 12.8 (Stanifer et al. 2018). Subsequent preliminary data from the heavily exposed Veneto, Italy, population also suggest a significant increase in kidney cancer mortality with PFAS exposure (Mastrantonio et al. 2018). Evidence is accumulating for PFAS as a cause of chronic disease and kidney cancer. Study designs must consider the peculiar PFAS excretion mechanics involved in and associated with kidney disease.

Reproductive and developmental outcomes

Exposure to PFOA impairs human sperm motility and sperm penetration into viscous media (Sabovic et al. 2020; Yuan et al. 2020) and is longitudinally associated with lower sperm concentration and count and higher adjusted levels of luteinizing and follicle-stimulating hormones in young men (Joensen et al. 2009; Vested et al. 2013; Song et al. 2018). Serum concentrations of PFAS are also cross-sectionally associated with deleterious markers of semen quality (Louis et al. 2015; Pan et al. 2019).

Legacy and emerging PFAS have been found in follicular fluid (Kang et al. 2020). They appear to alter endometrial regulation such as progesterone activity in young women (Di Nisio et al. 2020b) and possibly menstrual cycle length (Lum et al. 2017). Associations with menarche and menopause may be substantially due to reverse causation because menstruation is a route by which women eliminate PFAS (Dhingra et al. 2017), partially explaining why men have higher PFAS levels than women in the same communities. Women on birth control and who do not menstruate or with poor cyclicity because of age, activity level, or disease may have elevated PFAS levels in comparison with menstruating women. Exposure to PFAS has been associated with endometriosis in the United States and in China (Louis et al. 2012; Campbell et al. 2016; Wang B et al. 2017a), but the specific PFAS associated with this effect vary among studies.

Time-to-pregnancy (fecundity) studies provide indirect evidence of changes in fertility. Methodologic considerations include maternal and paternal age, parity (which in turn affects serum PFAS), and health status. Among 1240 women in the Danish National Birth Cohort, PFOS exposure was associated with decreased fecundity (median serum PFOS 35.5 ng/mL; Fei et al. 2009). Reverse causation may explain this finding because it is duplicated in parous, but not among nonparous, women (Whitworth et al. 2012; Bach et al. 2015). Prospective odds of actual infertility in the Maternal-Infant Research on Environmental Chemicals cohort ($n = 1743$) at low-dose exposures were associated with PFOA (geometric mean 1.66 ng/mL; odds ratio = 1.31, 95% CI 1.11–1.53) and PFHxS (odds ratio = 1.27, 95% CI 1.09–1.48; Velez et al. 2015). The reported fertility rate improved following water filtration in a PFAS-contaminated community (incidence rate ratio 0.73, 95% CI 0.69–0.77 prior to filtration) along with measures of birth weight (Waterfield et al. 2020).

Per- and polyfluoroalkyl substances reliably move across the placenta and enter breast milk (Gyllenhammar et al. 2018; VanNoy et al. 2018); serum PFAS levels in young children generally exceed maternal serum concentrations (Fromme et al. 2010; Papadopoulou et al. 2016; Eryasa et al. 2019). Population studies provide evidence that breastfeeding duration and milk quantity are adversely affected by PFAS exposure (Romano et al. 2016; Timmermann et al. 2017b; Rosen et al. 2018).

A systematic review reported that PFOA exposure was associated with a small decrease in infant birth weight; the meta-analysis estimated that a 1-ng/mL increase in PFOA was associated with an approximately 19-g reduction (95% CI −29.8 to −7.9 g) in birth weight (Lam et al. 2014). The authors noted similarities in experimental studies (Johnson et al. 2014; Koustas et al. 2014) and concluded that there was “sufficient” human and corroborative toxicology evidence of a detrimental effect of PFOA on birth weight (Johnson et al. 2014; Koustas et al. 2014; Lam et al. 2014). However, another meta-subpopulation analysis, focused on early pregnancy or the time shortly before conception, detected only a small and nonsignificant association, which was less subject to bias (Steenland et al. 2018a). Different approaches to the possible confounding role of shifting glomerular filtration rates in pregnancy can affect interpretations; evidence suggests this consideration can, at most, only partially explain associations of PFAS exposure to decreased birth weight (Interstate Technology and Regulatory Council 2020; Wikstrom et al. 2020). A recent review of mostly prospective cohort studies ($n = 24$ studies) noted PFAS associated with altered fetal and postnatal growth measures, such as lower birth weight. Many ($n = 22$) of the relevant studies suggest developmental and childhood immunomodulatory effects, whereas 21 studies concerning neurodevelopment were inconclusive (Liew et al. 2018). The authors of the review noted methodologic challenges of developmental and newborn epidemiology, including consideration of critical exposure windows for developmental effects, the effects of breastfeeding and parity on maternal PFAS levels, and the variety of possible mechanistic explanations for growth outcomes, such as disruption of glucocorticoid and thyroid hormone metabolism in utero (Liew et al. 2018). Recent Faroe Island studies report that prenatal PFAS effects on thyroid hormone status do not support a causal relationship (Xiao et al. 2020).

Review articles suggest that prenatal exposure to PFOA may increase risk of subsequent childhood adiposity, noting that steroid hormones, retinoid X receptor, and other pathways may be contributing to this effect (Halldorsson et al. 2012; Hall and Greco 2019). Prospective evidence supports this relationship in adults with a high risk of diabetes (Cardenas et al. 2017). However, some well-performed community studies do not support this outcome in adults or children (Barry et al. 2014; Martinsson et al. 2020).

Based on several preliminary findings, supported by longitudinal follow-up studies (Stein et al. 2009; Savitz et al. 2012; Darrow et al. 2013; Avanasi et al. 2016a, 2016b), the C8 Science Panel concluded that PFOA is probably linked to pregnancy-induced hypertension or preeclampsia. Population-level evidence implicating additional PFAS having this effect has included studies with longitudinal designs (Huang et al. 2019; Wikstrom et al. 2019; Borghese et al. 2020). Experimental support includes PFAS effects on human trophoblast migration in vitro (Szilagyi et al. 2020) and recent evidence of PFOA and GenX (or hexafluoropropylene oxide dimer acid) effects on mouse placenta, as well as excessive gestational weight gain (Blake et al. 2020). However, a recent longitudinal study did not find an association of PFAS with pregnancy-associated hypertension (Huo et al. 2020).

The possibility that circulating PFAS may reduce bone mineral density has been investigated. Cross-sectional and practical trial associations have been found in adults (Lin et al. 2014; Hu et al. 2019; Di Nisio et al. 2020a), and there is emerging longitudinal evidence from a mother and child pair study indicating that children may also be affected (Cluett et al. 2019).

Testicular cancer diagnoses are increasing steadily, a trend unrelated to improved detection (Cheng et al. 2018; Park et al. 2018). Most patients diagnosed (>90%) will be cured and die of other causes; mortality studies therefore provide little help in understanding disease risk factors. The C8 Science Panel detected longitudinal evidence for increased testicular cancer risk (1.35, 95% CI 1.00–1.79) for cumulative PFOA exposure (Barry et al. 2013). There are ample supportive data of testicular damage following PFAS exposure, including strong evidence of endocrine disruption; but the cell-specific associations are different in humans (germ cell) than the outcomes in rodents (stromal).

Per- and polyfluoroalkyl substances have deleterious effects on conception, pregnancy, and infant development. The underlying birth weight data are mostly supportive, although the subsequent growth and adiposity literature is mixed. The most sensitive reproductive and developmental outcomes are a topic of ongoing discussion.

Outcomes replicated across populations, such as perfluorocarboxylate (PFCA) and perfluorosulfonate (PFSA) exposures associated with down-regulation of immune response; increases in cholesterol, liver enzymes, and uric acid; alterations in thyroid hormone binding proteins; growth deficits; and effects on breast milk and lactation, indicate priority areas for understanding mechanisms and health implications.

CURRENT KNOWLEDGE OF PFAS TOXICITY IN EXPERIMENTAL MODELS

Animal studies have focused most intensely on PFOA and PFOS, using laboratory rodents and, more recently, zebrafish as models. Perfluoroalkyl acids of varied carbon-chain lengths as well as a few replacement chemicals with ether linkages in the carbon backbone (such as GenX and 3H-perfluoro-3-[(3-methoxy-propoxy)propanoic acid], or ADONA) have also been examined, with outcome profiles thus far generally consistent with legacy chemicals. The varying extent of responses is likely related to toxicokinetic disposition (excretion or half-life) and relative potency and affinity of the individual chemical for binding to receptor proteins. Some PFAS (i.e., PFHxS, PFOA, and PFNA) have longer half-lives in mice than rats and typically much longer half-lives in humans (Table 1). These differences in elimination kinetics complicate the cross-species evaluation of toxicity. In addition, some PFAS (such as PFOA and PFNA) exhibit a profound sex difference in the rate of chemical elimination and bioaccumulation in the rat: females eliminate them much faster than males (Table 1). Sex differences in half-lives, although important, are much smaller in humans and have a different explanation. The mouse also typically has more limited sex-based PFAS elimination differences, making this species more amenable for extrapolation to humans, especially for mechanistic and toxicity evaluations.

In general, human health effects associated with PFOA and PFOS exposure (described in section Current Knowledge of PFAS Toxicity in Humans) have also been reported in animal models: hepatic/lipid metabolic toxicity, developmental toxicity, immune suppression, tumor induction, endocrine disruption, and obesity. These findings are often derived from well-controlled laboratory experiments in more than one species using wide dose ranges that are often orders of magnitude higher than typical human exposure, to account for differences in half-life across species. Some of the phenotypic findings are supported by in vitro mechanistic investigation and/or molecular queries on target tissues. Our understanding of the toxicologic properties of PFAS other than PFOA and PFOS is notably less advanced and, in the case of emerging replacements and by-products, completely unexplored.

Hepatic and metabolic toxicity

In rodent studies, dose-dependent increases in liver weight, in hepatocellular hypertrophy associated with vacuole formation, and with or without increased peroxisome proliferation have been observed with a significant body burden of PFAS, especially for the most persistent and potent long-chain homologs. Hepatocyte proliferation, necrosis, and apoptosis are outcomes occurring at relatively low doses. This is also true for a new replacement chemical, GenX, which altered liver histopathology and function and increased apoptosis in mice and fish (Blake et al. 2020; Guillette et al. 2020). Correspondingly, transcriptional activation of mouse and, to a lesser extent, human PPARα-related genes in liver was detected in adult-exposed models; activation of other nuclear receptors such as PPARγ, constitutive androstane receptor (CAR), and pregnane X-receptor (PXR) has also been reported. These nuclear receptors, metabolic sensors that regulate lipid and glucose metabolism and transport and inflammation, tend to be more responsive in tissues of rodents than in humans (Wolf et al. 2012; Rosen et al. 2017). Recent work using developmental models reports that mitochondrial dysfunction is associated with hepatocellular hypertrophy in young adult mice (Quist et al., 2015) and that other fatty acid metabolism pathways are activated (Jones et al. 2003; Shabalina et al. 2016). Steatosis is also a common feature of PFAS chronic exposure in rodents. Exposure in rodent models typically decreases serum cholesterol, whereas elevations of circulating cholesterol levels have been reported in humans. The mode of action concerning serum cholesterol is debatable. For example, PFOA exposure increased liver weight, increased liver enzymes, and led to persistent histopathological changes (particularly damage to the bile duct) in livers of wild-type and PPARα-null rodent strains (reviewed in Division of Science and Research, New Jersey Department of Environmental Protection 2019). Many of these effects are reversible on cessation of PFAS exposure, and this observation has been interpreted by some as evidence of “adaptive” responses to exposure. However, this reversibility is irrelevant to ongoing environmental PFAS exposure (for instance, from drinking water) because exposure will persist until contamination is remediated. In summary, there is a strong confluence of animal toxicology and histology and human population data that PFAS disrupt hepatic metabolism and lead to lipid accumulation in liver, although the mechanism(s) is unclear. Effects on bile acid metabolism, mitochondrial perturbation, and cholestatic mechanisms deserve further investigation at human-relevant exposures.

Reproductive and developmental toxicity

Only a few reproductive toxicity studies of males and females are available, primarily focusing on long-chain PFAS. Profound developmental toxicity has been described following gestational and lactational exposure to PFOS, PFOA, and PFNA in mice (Thibodeaux et al. 2003; Lau et al. 2006; Das et al. 2015) and in mice and rats gestationally exposed to GenX (Conley et al. 2019; Blake et al. 2020). Neonatal morbidity and mortality were seen with exposure to high doses of legacy PFAS; growth deficits and developmental delays were noted in offspring exposed to lower doses. Evidence of lactation impairment was seen in mice at doses of 5 mg PFOA/kg body weight (White et al. 2007), leading to increased offspring mortality (Lau et al. 2006); recent studies have indicated a role of placental dysfunction in these adverse developmental outcomes (Blake et al. 2020). Deficits of mammary gland development were also observed in mice exposed to PFOA (doses of 1 mg/kg body wt and lower) during gestation, which persisted into adulthood, although these exposure levels did not alter body weight, lactational function, or neonatal growth of offspring (F1 or F2 mice; Macon et al. 2011; White et al. 2011b; Tucker et al. 2015). Systematic reviews support a relationship between in utero exposure to PFOA and PFOS and reduced fetal growth in animals and humans, and the relationship between PFOA and reduced fetal growth in mice was recently validated (Koustas et al. 2014; Blake et al. 2020). Also, PFAS are reported to have reproductive effects such as ovulation failure in mice (Zhang Y et al. 2020).

Immunotoxicity

A few long-chain PFAS (PFOS, PFOA, PFNA, and PFDA) have been shown to alter immune status in rodents and non-human primates. Effects are predominantly immunosuppressive and include reductions in thymus and spleen weights and associated immune cell populations, in numbers of circulating immune cells, in certain aspects of innate immunity (i.e., natural killer cell cytotoxicity), in infectious disease resistance, and in antibodies produced in response to an antigen (i.e., analogous to the vaccine response in humans). In their 2018 draft *Toxicological Profile for Perfluoroalkyls*, the US Agency for Toxic Substances and Disease Registry (ATSDR) noted changes to the aforementioned immune parameters observed in experimental rodents exposed to PFOA, PFOS, PFNA, PFHxS, PFDA, perfluorobutanesulfonic acid (PFBS), or perfluorobutanoic acid (PFBA; Agency for Toxic Substances and Disease Registry 2018). The US National Toxicology Program conducted a systematic review of the immunotoxicological literature for PFOA and PFOS and concluded that PFOA and PFOS were presumed to be immune hazards to humans based on a high level of evidence for suppression of antibody responses in experimental animals and a moderate level of evidence for suppression of antibody responses in humans (National Toxicology Program 2016). The ATSDR (Agency for Toxic Substances and Disease Registry 2018) also included a decreased antibody response to vaccines (PFOA, PFOS, PFHxS, and PFDA) and increased risk of asthma diagnosis (PFOA) among the list of adverse health effects in PFAS-exposed humans. Reduction in the antibody response to a vaccine, an adaptive immune function, is a well-accepted measure of immunotoxicity, is consistent with the mode of action for the effects of fatty acids on immune system function (Fritsche 2006), and is compelling evidence that the immune system is a sensitive target of PFAS.

Tumor induction

Per- and polyfluoroalkyl substances are not known to be directly mutagenic; PFOA, PFOS, and other tested PFAS show little or no evidence for induction of gene mutation, clastogenicity, or aneuploidy in vitro or in vivo by a direct mode of action (see EFSA Panel on Contaminants in the Food Chain [2020] for details). There is evidence that PFAS can induce DNA damage, such as strand breaks, and other genotoxic effects, secondary to oxidative stress (EFSA Panel on Contaminants in the Food Chain 2020). This occurs at concentrations or doses that are high relative to human environmental exposures to PFAS, and the mechanism is such that their dose-response will be sublinear. Hence, PFAS are unlikely to be of mutagenic concern in exposed populations.

In adult-exposed rodents and fish, PFOA and PFOS have been shown to induce tumors. Liver adenomas, pancreatic acinar cell tumors, and testicular Leydig cell adenomas have been detected in rats treated chronically with PFOA (IARC Working Group on the Evaluation of Carcinogenic Risks to Humans 2017) as well as its replacement, GenX (Caverly Rae et al. 2015). Following gestational and chronic exposure to PFOA, 58% of male rats demonstrated pancreatic tumors at the lowest dose administered (National Toxicology Program 2020b). This finding has spurred Minnesota and California policymakers to consider cancer as an endpoint in risk assessment, whereas the European Food Safety Authority (EFSA Panel on Contaminants in the Food Chain 2020) has the opinion that there is not adequate evidence for a link between exposure to PFAS and cancer risk in humans. This “tumor triad” profile has been associated with the PPARα-mediated molecular signaling pathway in rats exposed to high doses of PFAS. Consequently, liver tumors involving this mode of action are not considered relevant to humans at equivalent PFAS exposures (Post et al. 2017). The human relevance of PPARα-mediated pancreatic tumors in rodents remains to be determined. Liver lesions evident in PPARα-null mice exposed to PFOA during pregnancy and lactation (Filgo et al. 2015) suggest a non-PPARα-mediated liver response. Induction of liver tumors mediated by estrogen receptor (ER) activation has also been reported in fish (Tilton et al. 2008), and several non-PPARα-mediated hypotheses, including increased reactive oxygen species formation, oxidative stress, and mitochondrial dysfunction; decreased tumor cell surveillance by the immune system; and diminished gap junction cellular communication, are documented (IARC Working Group on the Evaluation of Carcinogenic Risks to Humans 2017; New Jersey Drinking Water Quality Institute Health Effects Subcommittee 2017).

Endocrine disruption

The primary evidence for the endocrine-disrupting potential of PFAS involves induction of hypothyroxinemia and reduction of serum testosterone in rats. An early review of PFAS endocrine-disrupting properties in humans concluded that the “thyroid may be one axis significantly affected by PFOA exposure while the animal toxicology literature is less certain due to technical issues” (White et al. 2011a).

The effects of PFAS on thyroid hormone status detected in animal studies differ from classical hypothyroidism, in that reduction of circulating total thyroxine is not accompanied by a compensatory increase of TSH. A possible mechanism for these effects may be related to the propensity of protein binding of legacy PFAS, which could lead to displaced total thyroxine binding to its carrier proteins (transthyretin and thyroxine-binding globulin). Human population studies augment animal data showing that PFAS interact with thyroid hormone binding proteins (Berg et al. 2015; Ren et al. 2016; Zhang J et al. 2016a), one of several mechanisms by which PFAS can perturb feedback relationships between free thyroid hormone available to cells (free total thyroxine) and the hypothalamic-pituitary axis. Some estrogenic effects of PFAS have also been illustrated by in vitro studies, although there is no evidence of direct transactivation of estrogen, androgen, or glucocorticoid receptors (Behr et al. 2018, 2020b).

The evidence for PFAS affecting ER signaling in humans and animals is mixed. Although studies have identified some PFAS as being without estrogenic activity (Behr et al. 2018; Borghoff et al. 2018; Gogola et al. 2019), others suggest an ability of PFAS to modulate or even activate ER-mediated effects (Benninghoff et al. 2010; Kjeldsen and Bonefeld-Jørgensen 2013; Wang et al. 2018; Bjerregaard-Olesen et al. 2019; Qiu et al. 2020), with some effects only observed in aquatic organisms (Wei et al. 2009; Chen et al. 2016, 2018).

Microarray analyses of human primary hepatocytes confirmed that PFOA activated the ER pathway (Buhrke et al. 2015).

Neurotoxicity

Potential adverse effects of PFAS on the nervous system and functions have not been widely investigated. A few studies reported neurotoxicity of PFOS, PFHxS, and PFOA in cell culture systems (Slotkin et al. 2008), as well as altered behavioral responses (Goulding et al. 2017) and deficits in learning and memory ability in rodents (Viberg et al. 2013). In contrast, no significant developmental neurotoxic effects were seen from prenatal exposure to PFOS in USEPA guideline-based studies with rats (Butenhoff et al. 2009).

Obesity

Numerous cell-based assays in human and mouse pre-adipocytes and animal studies with and without high-fat diets have consistently shown that some PFAS have the potential to increase lipid production by adipocytes and fat pads (van Esterik et al. 2016). Exposure of pregnant mice to low doses of PFOA produced obesity in young adult female offspring (Hines et al. 2009; van Esterik et al. 2016), a finding that was re-capitulated in Danish women exposed in utero to PFOA (Halldorsson et al. 2012). Both PFOA and GenX increased weight gain of pregnant mice (Blake et al. 2020), an effect also seen in women during pregnancy (Ashley-Martin et al. 2016), although discordant results have been reported in other studies (Barry et al. 2014; Ngo et al. 2014). These apparently disparate findings in experimental models may be associated with differences among mouse strains examined, exposure periods, statistical methodology, and/or the rodent diets used.

There are specific differences in human and rodent health outcomes that deserve further investigation: 1) cholesterol metabolism, 2) thyroid effects, 3) mode of action for liver effects (different or same), and 4) kidney transporter or other mode of action leading to large differences in half-life. However, species concordance in the 6 human health effects discussed in the present review supports a weight of evidence for these effect for the handful of extensively studied PFAS.

Human health advisory and guidance values for a few PFAS have been issued to date by the USEPA, the ATSDR, several individual state environmental agencies or health departments, as well as regulatory agencies in Canada and Europe that are largely (but not exclusively) based on toxicological findings in animal models. However, risk-assessment scientists have not reached consensus in selecting a singular apical endpoint as the basis for a point of departure for assessments. Three toxicological features of PFAS that have been commonly highlighted, based on their sensitivity (low dose effect), strength of evidence (robust corroborating studies with mechanistic support for human relevance), and corresponding findings noted in epidemiological investigation, are hepatotoxicity (and alterations in lipid metabolism), developmental toxicity, and immunotoxicity. It should be noted that apical endpoints that drive risk assessments often differ among individual PFAS, perhaps highlighting the complexity of these chemicals and the family of PFAS, in general.

IMPORTANCE OF TOXICOKINETICS IN UNDERSTANDING PFAS TOXICITY

Species and sex differences

Few of the substantial number of structurally diverse PFAS have been tested for toxicological effects. Some available toxicological information has come from studies in animals, where marked species and (in rat) sex differences in half-life for some PFAS (Table 1) have been observed and the relevance to humans is uncertain. These differences are due to toxicokinetic and toxicodynamic factors. There are also differences in mean PFAS serum levels between men and women in the same communities. Children may have elevated serum levels compared to parents, even with the same exposures (Emmett et al. 2006; Daly et al. 2018; Graber et al. 2019), for reasons relating to transplacental transfer, breastfeeding, and body mass (Emmett et al. 2006; Daly et al. 2018; Graber et al. 2019; Blake et al. 2020). Transplacental transfer of PFAS confers a substantial burden to the newborn infant. Because the infant has a smaller overall mass and blood volume, PFAS are concentrated, increasing PFAS per volume (Koponen et al. 2018). In addition, transfer of PFAS is common through lactation, and the longer a child breastfeeds, the higher the body burden (Gyllenhammar et al. 2018; VanNoy et al. 2018).

Effects of comorbidity on PFAS toxicokinetics

Factors affecting renal function can influence PFAS toxicokinetics. As discussed, opposing types of causation should be considered. Human toxicokinetics appear to vary bidirectionally with changing renal function, leading to nonmonotonic dose-response relationships and, depending on the study goal, possibly to errors in estimating disease associations. As progress is made in the field of PFAS toxicokinetics, new chemistries may have different clearance factors and nuances that vary by PFAS group or structures, and that will need to be investigated to accurately model half-lives in different exposure subgroups.

Sources of information on toxicokinetics in humans: strengths and limitations of studies

Some PFAS half-life data in humans were obtained from retired industry workers, particularly those who worked with PFOS, PFOA, and PFHxS (Olsen et al. 2007). Since then, these estimates have been modified slightly or confirmed with longitudinal data and modeling from contaminated communities once uncontaminated water options were provided (Bartell et al. 2010; Li et al. 2018). Other contemporary PFAS estimates are derived from biomonitoring studies of production workers, blood donors, study participants, and/or occupationally exposed cohorts (Olsen et al. 2009, 2017; Russell et al. 2013; Zhang et al. 2013). Some caution must be taken in using these data because variables affecting PFAS clearance may not be taken into consideration (age, sex, menstruation, disease, and medication status) and may contribute to confounding.

The challenge in determining a reliable human half-life in these types of studies is that exposure does not end with a clean water source, retirement, or a change of job and that continued exposures vary over potential depuration periods. Model components may also vary in subclasses. Children (small blood volumes and a large fraction of exposures comes from drinking), pregnant women (large increase in blood volume and water intake), parous women (transfer to fetus and breast milk), and athletes (water intake elevated) are examples of subpopulations with expected variation in half-life compared to adult men (Post et al. 2017). There will be more human estimates of PFAS forthcoming that involve variations in half-life (Post et al. 2017). Realistic computational modeling can help, so long as it clearly characterizes exposures and applicable populations. The continued goal should be to provide predictive values for those PFAS lacking actual measurements, based on chemical structures and trusted physiological parameters.

Physiologically based pharmacokinetic/toxicokinetic modeling in different-aged populations

In the blood and other tissues, PFAS toxicokinetics are influenced by their interactions with proteins (Andersen et al. 2006; Katakura et al. 2007; Nakagawa et al. 2008; Weaver et al. 2009; Figure 2). Certain toxicokinetic features are saturable, and thus dosing in toxicokinetic studies is of profound importance. Studies of renal reabsorption mechanisms in mammals show that reduced activity of transporters such as organic anion transporting polypeptide 1a1, through inactivation (e.g., genetic manipulation, castration, treatment with estrogen) or by saturation at increasing doses, leads to substantial reductions in half-lives of PFOA and PFOS (Andersen et al. 2006; Nakagawa et al. 2008; Weaver et al. 2009; Yang et al. 2009).

These protein-associated toxicokinetic processes were recently incorporated into a model for PFOA in the male Sprague-Dawley rat (Cheng and Ng 2017), which provides a useful platform to explore how changes in protein interactions might affect estimates of PFAS half-life (Figure 3). At high doses, it is typical to see clear biphasic behavior with rapid initial clearance, during which the serum half-life appears to be shorter especially at high enough doses that processes such as renal reabsorption are saturated, followed by a much longer tail (Figure 3A). In a similar fashion, the magnitude of internal dose and rate of serum clearance can be profoundly influenced by proteins known to bind PFAS, such as serum albumin (Figure 3B). Increasing and decreasing the extent of reabsorption in the kidney increases and decreases the serum half-life, respectively (Figure 3C). Finally, the effect of saturating reabsorption is magnified when the half-life is longer because of increased serum binding (Figure 3D). In this case, taking an initial slope to calculate the serum half-life at high doses would lead to a profound underestimation.

Differences in protein expression, circulating levels, and even protein type across populations, sex, and species could lead to important species and sex differences in PFAS biological half-lives (Han et al. 2012); such differences should be investigated and taken into account in the extrapolation to human equivalent doses. Because expression of proteins may change at different life stages, clearance factors and toxicokinetics may also change.

Given the large number of species-, sex-, and age-specific differences that have been observed, coupled with the lack of data for many PFAS, the parameterization of complex physiologically based toxicokinetic models remains a persistent challenge. Therefore, lower-resolution models (e.g., one-compartment or few-compartment models) may be more appropriate for species and settings where insufficient data are available for reasonably accurate parameterization. Alternatively, in silico and in vitro methods are under development that could aid in parameterization in the absence of in vivo data, as discussed in the section New approaches for developing PFAS toxicity information.

SO MANY PFAS, SO LITTLE TIME: ACCELERATING THE PACE OF DISCOVERY

Importance of determining mode of action and adverse outcome pathways

Information on modes of action and/or adverse outcome pathways (AOPs) is invaluable in 1) establishing human relevance of experimental evidence, 2) assessing causality in epidemiological studies, 3) applying “read-across” to PFAS for which there is little toxicological information, 4) assessing risks from mixtures, 5) guiding development and interpretation of new approach methodologies, 6) informing the development of biomarkers in epidemiologic investigation, and 7) identifying potentially vulnerable subpopulations and life stage-specific effects (Meek et al. 2014; LaLone et al. 2017). Verified modes of action and AOPs can inform risk assessment based on intermediate effects and enable development of new methodology-based approaches to assess PFAS safety (Meek et al. 2014).

Postulated modes of action/AOPs for PFAS

Mechanistic studies have been performed on only a few PFAS. These have been shown to activate a range of putative molecular initiating targets, among which are the nuclear receptors PPARα, PPARγ, PPARβ/δ, CAR, PXR, liver X receptor α, and ERα (Bijland et al. 2011; Bjork et al. 2011; Rosen et al. 2017; Li et al. 2019). However, modes of action verified by agreed procedures (World Health Organization 2020) have been established for few reported effects of PFAS, and those that have been interrogated involve activation of PPARα and, at higher doses, CAR as molecular initiating events (Klaunig et al. 2012; Rosen et al. 2017). Several AOPs involving these molecular targets are in various stages of development (Organisation for Economic Co-operation Development 2020), but few have been endorsed by the OECD following its agreed procedures (Organisation for Economic Co-operation Development 2017). Demonstration of receptor activation alone is insufficient to establish involvement of a mode of action or AOP in an observed effect, for which an overall weight-of-evidence approach is necessary (World Health Organization 2020).

Andersen et al. (2007) provide a useful, albeit dated, review of possible PFAS modes of action. Established modes of action are restricted largely to the liver and include species-specific hepatic hyperplasia and liver tumors (Butenhoff et al. 2012; Elcombe et al. 2012; Corton et al. 2018). Available studies on PFBS, PFHxS, perfluorohexanoic acid, PFNA and PFDA suggest that they share molecular targets with similar consequences, albeit with differences in potency, in part due to differences in their excretion and protein-interaction kinetics (Zeilmaker et al. 2018). However, studies in vitro have established intrinsic differences in potency among PFAS analogues. Potency in activating PPARα showed some relationship with PFAS chain length (Wolf et al. 2008). A mode of action or AOP provides a causal chain of key events between chemical exposure and outcome. The established modes of action for PFOS and PFOA provide a causal explanation for development of liver tumors observed in rodents on exposure to these compounds, through activation of PPARα, and the possible relevance to humans. However, this does not mean that other effects of PFAS are due to activation of PPARα or that other pathways might not lead to liver tumors in humans, such as secondary to the primary effect of steatosis.

Until recently, there has been little study of modes of action/AOPs for effects of PFAS other than hepatic outcomes in rodents, particularly for critical effects, such as immunosuppression and developmental toxicity, and from PFAS other than PFOS and PFOA (EFSA Panel on Contaminants in the Food Chain 2020; Temkin et al. 2020). The ability of various PFAS to interact with and modify lipid metabolism is, however, an intriguing hypothesis (Xu et al. 1999; Jones et al. 2003; Andersen et al. 2007; Tan et al. 2013; Pouwer et al. 2019). Other putative molecular initiating/key events for PFAS, in addition to nuclear receptor activation, include gap junctional inhibition to disrupt cell-cell communication, mitochondrial dysfunction, interference of protein binding, partitioning into lipid bilayers, oxidative stress, altered calcium homeostasis, and inappropriate activation of molecular signals controlling cell functions. Many of these effects are consistent with a nonspecific action of PFAS on the cellular lipid membrane (Spector and Yorek 1985; Bourre et al. 1989; Dodes Traian et al. 2012; Casares et al. 2019). However, these alternative events lack robust evidence to support a specific pathophysiological role in the multifaceted effects of PFAS. A better characterization of the modes of action/AOPs for PFAS toxicities remains an important area of future investigation, necessary to improve our understanding of PFAS impacts on human health.

At present, there is insufficient evidence to determine which of, and to what extent, these molecular interactions play a pathophysiological role in observed adverse outcomes of PFAS (Michigan PFAS Science Advisory Panel 2018). Hence, there is a need to integrate such mechanistic information into a weight-of-evidence framework, first by establishing the mode of action or AOP linking a proposed chain of key events to an adverse outcome and then by demonstrating that at human exposure levels of PFAS the established AOP or mode of action is causal in the adverse outcome observed. The substantial advantage offered by such an approach is the ability to read across from representative members of appropriate PFAS groupings, based on quantitative information from new approach methodologies and exposure estimates. Hence, better characterization of the modes of action/AOPs for PFAS toxicities remains a critical area of future investigation and will allow us to understand which adversely PFAS-modified pathways must be interrogated prior to new chemicals joining this class. Predicting PFAS activity in the body should be the goal prior to approving novel PFAS for use.

New approaches for developing PFAS toxicity information

When it comes to determining which PFAS should be prioritized for further testing, there are too many chemicals, even in one subclass, for traditional approaches. Numerous creative and high-throughput methodologies are being developed and tested to provide valuable data on PFAS with no toxicity data.

Collaborative approaches.—Problem formulation and approach must be guided by available equipment, funds, and technical staff, and important principles: 1) What biological activity and toxicology information can be generated in a *responsive time frame*? 2) Can this information be used to make public health decisions? 3) What are appropriate tools to bring to this problem (platforms, species/sex of cells used, metabolic competency of the model system, and data analysis)? 4) How do we organize, and what are the best mechanisms to report useful biological activity/toxicological information?

Developing "how" to evaluate potential health effects of new PFAS requires some thought to PFAS heterogeneity. Although subclass names have been suggested by several investigators (Buck et al. 2011; Wang Z et al. 2017; Sha et al. 2019), there is still disagreement on those groupings. In addition, half-lives and biological persistence are not predictable based on structure, and exposure routes may be complex. Given that traditional approaches to generate toxicity information are resource-intensive, new approach methodologies, which may include in vitro high-throughput toxicity screening and toxicokinetic testing, will be needed to inform further (in vivo) testing of PFAS.

One example of how agencies/institutes are collaborating to prioritize a list of PFAS needing further study is the REACT Program (Responsive Evaluation and Assessment of Chemical Toxicity). Scientists from the USEPA and the National Institute of Environmental Health Sciences (NIEHS) National Toxicology Program have joined forces to determine if read-across approaches would work. Essentially, they will use existing data for a data-rich substance (the source, e.g., PFOA or PFOS) as an anchor for a data-poor substance (the target, a novel PFAS), which is considered similar enough to the source substance to use the same data as a basis for the safety assessment. For example, the US National Toxicology Program 28-d PFAS or chronic PFOA data set (National Toxicology Program 2020c) could be used as an anchor. The goal is to group PFAS by biological activities and then use in vitro to in vivo extrapolation data and models to estimate oral equivalent exposures for PFAS. For example, multiple biological endpoints (Table 2) were chosen to generate data on 150 PFAS (Patlewicz et al. 2019), representing several structural subclasses for use in read-across.

Selecting assays shown in Table 2 based on PFOA and PFOS health effects covers a broad range of biology. However, because of the structural diversity of PFAS, biological activity of subclasses of PFAS may be missed; but this can be addressed in 2 ways. First, using transcriptomics as a screen, similar and unique pathways altered by different PFAS can be identified. Second, structure-activity relationships may predict potentially missing biological activities. As an example, Leadscope model predictions conducted at the NIEHS predicted biology that was covered in assays already chosen for evaluation, which increased confidence in the approaches chosen. Because model predictions are only as robust as data sets from which they are generated, these outputs should be used to identify assays for screening efforts and not as synonymous with toxicities induced by PFAS. Ultimately, the REACT program aims to prioritize PFAS for additional targeted testing and follow-up with in vivo studies as needed.

Molecular dynamics and protein interactions.—Advances in computational tools, many developed for drug discovery, allow environmental and public health researchers to better anticipate some impacts of emerging contaminants even in the absence of substantial experimental data (Rabinowitz et al. 2008). For example, molecular docking and molecular dynamics to predict strengths of interactions between biomolecules and contaminants can be an in vitro screening tool for assessing legacy and emerging PFAS for bioaccumulation potential, to identify potential sites of toxic action (Salvalaglio et al. 2010; Ng and

Hungerbuehler 2015; Cheng and Ng 2018; Li et al. 2019) and to gain insights into toxic mechanisms (Sheng et al. 2018). Relatively strong binding with particular proteins (e.g., serum albumin, liver fatty acid binding protein) has already proven useful in correlating PFAS structure with potential for bioaccumulation (Ng and Hungerbühler 2014; Cheng and Ng 2017). Tools including molecular docking and molecular dynamics can correlate relative binding affinities of emerging PFAS with these target proteins and subsequently compare with affinities of legacy chemicals with known bioaccumulation potentials, thus providing a first-tier rapid screening mechanism (Luebker et al. 2002; Cheng and Ng 2018).

The use of fluorinated substances in pharmaceutical products has led to an unexpected data source for discovery of structural features in PFAS associated with various types of bioactivity. These data were recently used to train machine learning models to predict potential bioactivity for thousands of untested PFAS (Cheng and Ng 2019). Classification approaches such as these serve as preliminary screening tools for identifying PFAS as a first step in a tiered assessment when detailed mechanistic information is not available.

Addressing mixtures.—Based on their potential for complex exposure patterns, PFAS are a mixtures issue. Communities with water-monitoring programs reporting PFAS concentrations demonstrated that they are exposed to mixtures of PFAS. This mixture may be from one or more point sources releasing multiple PFAS and/or PFAS by-products into the air and water, such as a Chemours plant in North Carolina, and suggest that exposures may be substantial (McCord and Strynar 2019). However, numerous other PFAS sources are known to impact community exposure to PFAS mixtures, such as landfill leachate, biosolids recycling, and aqueous film-forming foam contamination of drinking water sources, among others (Sunderland et al. 2019; Solo-Gabriele et al. 2020). Aqueous film-forming foam and other mixtures evident in drinking water, food packaging, health and beauty products, and food-based sources are often poorly characterized (Sunderland et al. 2019; Susmann et al. 2019).

Discussions on whether PFAS may be addressed using a relative potency framework or toxic equivalency factor approach are ongoing. Substances could be grouped by bioaccumulation and persistence (toxicokinetics), function (biology), molecular initiating events, with potency factors derived from several assays, or subclass (structural similarity).

SPECIAL CONSIDERATIONS IN FUTURE STUDY DESIGNS

Future epidemiological studies

Future human studies need to characterize immune outcomes including (and not limited to) immune effects from exposure in early pregnancy and possible roles of PFAS in initiating allergic and autoimmune processes, conditions for which a dose response is hard to predict. Interactions of immune pathways with liver and lipid toxicity deserve additional consideration.

Liver and lipid studies have reasonably characterized associations between PFAS and effects and should now address why and what to do about it. Characterization of possible a priori susceptibility, such as in the obese, is important. Human and animal lipid data suggest that future experimental studies should focus on mitochondrial toxicity, alterations in bile acid metabolism, cholestasis, and resultant steatosis. These outcomes are already known to be associated with altered serum lipids, liver enzymes, and uric acid in the human population regardless of PFAS (Cohen and Fisher 2013; Sattar et al. 2014; Arguello et al. 2015; Jensen et al. 2018).

Studies of human kidney markers related to PFAS exposures illustrate the importance of understanding physiology to inform study design choices and reasonable interpretations. These substances have complex excretion mechanics that vary with dose, state of the healthy or progressively diseased kidney, as well as a potentially additional causative effect on kidney disease outcome(s). Appropriate definition of biological and mechanistic targets and more precise investigation of PFAS subclasses will better inform study designs and research questions. For example, consistent reports of disrupted cholesterol metabolism should prompt mechanistic studies evaluating effects on steroid hormones that may influence cancer, fecundity, lactation, and developmental signals seen in human population data. More attention could be given to effects of PFAS on the hypothalamic-pituitary-gonadal axis and then reconsidered based on life stages.

The history of long-chain PFAS studies indicates that collaborative team approaches featuring clinical, epidemiologic, computational modeling, and laboratory toxicological expertise are needed. Future population designs and more sensitive analytical methodologies should address replacement chemicals, typically found as mixtures; study designs must account for shorter PFAS half-lives and unpredictable PFAS detection in exposed individuals/communities. Innovative use of biomarkers in specifically designated risk subpopulations (obesity, immune) will likely be important.

Sex differences, nonmonotonic dose responses, sensitive subpopulations

Although serum-level differences exist between men and women similarly exposed to individual PFAS, sex-dependent differences in half-life have not been reported in human populations for short-chain (PFBS, PFBA) or long-chain per-fluoroalkyl acids thus far (Li et al. 2017b). Perhaps the half-life differences between the sexes is similar to interindividual variability and cannot be detected above background, or studies deriving data sets used to model half-lives were not designed to detect sex differences (convenience sampling or workers were mostly male, etc.). However, sex-specific elimination half-lives are defined (Table 1) for some PFAS in rodent models. In addition, developmental exposure studies in experimental models have consistently shown effects at lower doses than adult-only exposures and should be given priority in testing replacement chemicals. In vitro and alternative models that capture developmental susceptibility are encouraged. In summary, care should be taken in testing replacement PFAS in rodent or alternative (cell-based or zebrafish, for example) models to consider 1) the possibilities of sex-based differences in elimination half-lives, 2) dose range used (to include human relevant exposures), 3) life stage represented in the model system, and 4) variability of the response to enable the use of data generated for risk assessment.

Future experimental model studies

Experimental rodent studies have been essential in confirming PFAS health effects (liver and thyroid disease, lipid homeostasis), even when effects were not identical to those in humans; in some cases, novel targets (mammary and immune changes) were identified in animals. Future animal, cell-based, and high-throughput toxicity screening should enhance transparency in reporting to include blinded dose allocation, reporting of all data, adherence to Animal Research Reporting In Vivo Experiments (ARRIVE) guidelines (Kilkenny et al. 2010), and dose ranges that approach human relevance (adjusted to reflect the differences in elimination between species and potentially chronic exposures) so that they suitably inform systematic reviews that may be used in chemical regulation.

Model selection for health effects evaluation is critical. An appropriate model should be sensitive, be susceptible to the outcome(s) of interest (obesity, immune), and produce outcomes that will inform human health effects. Alternative research models, such as transgenic mice, zebrafish, developmental models for most affected target tissues, and diet-challenged designs in susceptible rodent strains, will strengthen our knowledge of PFAS-related health effects. Validation of fish neurobehavior models to inform mammalian, including human, developmental responses is needed.

Finally, advanced human cell-based platforms—that have been validated for relevant outcomes in humans—will facilitate concurrent screening of larger numbers of PFAS, but bioavailability of PFAS in the culture system needs to be understood because binding to media proteins or labware, the instability of some PFAS in some vehicles, and altered metabolism may exist in some cases (Gaballah et al. 2020; Liberatore et al. 2020).

Future alternative approaches

One way to determine the toxicity of the large number of PFAS compounds currently used in commerce is to develop quantitative structure-activity relationships (QSAR). Such QSAR attempt to define relationships between a PFAS compound structure with a specific biological activity or response that identifies or is a biomarker for toxicity. Few data are available for receptor binding of PFAS, mainly limited to a few PFCAs and PFSAs; and even between carboxylates and sulfonates of similar chain length substantial differences have been observed (Cheng and Ng 2017, 2018). If there are substantial differences between perfluoroalkyl carboxylic and sulfonic acids, which differ only in their acid head group, construction of successful QSAR for the large and diverse class of all PFAS will be particularly challenging. Several QSAR may be developed, each predictive of toxicity of a distinct class or subclass of PFAS, based on a unique functional moiety or other feature. Although this brings additional challenges in finding sufficient data for QSAR training and validation, big data approaches, such as the recently developed machine learning models to predict PFAS bioactivity (Cheng and Ng 2019), show promise for advancing these computational approaches at the screening level.

For example, it may be determined by affinity for receptor-specific binding and nonspecific interactions with cellular membranes that the specific toxic effect exhibits a multiphasic dose response reflecting 2 potential modes of action. In addition, the critical effect may change with levels of PFAS exposure. Add to this that people are typically exposed to PFAS mixtures, each of which may have a different affinity for a binding site and ability to impact cellular membrane fluidity, and the potential to predict PFAS toxicity becomes extremely complicated. In the foreseeable future, we may be limited to assessing PFAS toxicity using high-throughput assays designed to inform regulators as to the relative toxicity of PFAS mixtures or compounds. Such approaches are suited to the use of artificial intelligence (i.e., machine learning approaches) that integrate data from multiple sources to identify bioaccumulation potential, relevant pathways triggered, protein binding affinities, and modes of action involved in the development of individual and mixture toxicity of PFAS.

The utility of any future approach to determining PFAS toxicity must consider tissue-specific modes of action. Such an approach may rely on molecular interactions with specific binding sites on enzymes/storage/transport proteins or the nonspecific ability to alter cell membrane fluidity by which membrane-bound protein activities are altered within a particular organ/system. Regardless of the mode of action, model, and/or simulation, the predictive result should be biologically plausible and represent dose-effect responses across species.

CONCLUSION

Future research on the health effects of replacement PFAS and mechanistic studies on legacy PFAS must apply “lessons learned” such as those highlighted in the present review. There are only a handful of PFAS with enough health effects data for use in decision-making, as evidenced by state-led standard setting. There are numerous health effects reported for those PFAS tested, which sets this family of chemicals apart from many others and elevates the need for precautionary action. With hundreds of PFAS lacking health effects data, translational research teams using innovative methodologies and carefully designed studies will be critical to our state of knowledge on PFAS-related health effects and our enhanced strategies for informing risk assessment of this large family of chemicals.

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REFERENCES

Abraham K, Mielke H, Fromme H, Volkel W, Menzel J, Peiser M, Zepp F, Willich SN, Weikert C. 2020. Internal exposure to perfluoroalkyl substances (PFASs) and biological markers in 101 healthy 1-year-old children: Associations between levels of perfluorooctanoic acid (PFOA) and vaccine response. Arch Toxicol 94:2131–2147. [PubMed: 32227269]

Agency for Toxic Substances and Disease Registry. 2018. Toxicological profile for perfluoroalkyls. US Department of Health and Human Services, Washington, DC. [cited 2020 July 13]. Available from: https://www.atsdr.cdc.gov/toxprofiles/tp200.pdf

Alderete TL, Jin R, Walker DI, Valvi D, Chen Z, Jones DP, Peng C, Gilliland FD, Berhane K, Conti DV, Goran MI, Chatzi L. 2019. Perfluoroalkyl substances, metabolomic profiling, and alterations in glucose homeostasis among overweight and obese Hispanic children: A proof-of-concept analysis. Environ Int 126:445–453. [PubMed: 30844580]

Andersen ME, Butenhoff JL, Chang S-C, Farrar DG, Kennedy GL Jr, Lau C, Olsen GW, Seed J, Wallace KB. 2007. Perfluoroalkyl acids and related chemistries—Toxicokinetics and modes of action. Toxicol Sci 102:3–14. [PubMed: 18003598]

Andersen ME, Clewell HJ, Tan Y-M, Butenhoff JL, Olsen GW. 2006. Pharmacokinetic modeling of saturable, renal resorption of perfluoroalkylacids in monkeys—Probing the determinants of long plasma half-lives. Toxicology 227:156–164. [PubMed: 16978759]

Andersson EM, Scott K, Xu Y, Li Y, Olsson DS, Fletcher T, Jakobsson K. 2019. High exposure to perfluorinated compounds in drinking water and thyroid disease. A cohort study from Ronneby, Sweden. Environ Res 176:108540. [PubMed: 31252203]

Arguello G, Balboa E, Arrese M, Zanlungo S. 2015. Recent insights on the role of cholesterol in non-alcoholic fatty liver disease. Biochim Biophys Acta 1852:1765–1778. [PubMed: 26027904]

Armstrong LE, Guo GL. 2019. Understanding environmental contaminants' direct effects on non-alcoholic fatty liver disease progression. Curr Environ Health Rep 6:95–104. [PubMed: 31090041]

Ashley-Martin J, Dodds L, Arbuckle TE, Morisset AS, Fisher M, Bouchard MF, Shapiro GD, Ettinger AS, Monnier P, Dallaire R, Taback S, Fraser W. 2016. Maternal and neonatal levels of perfluoroalkyl substances in relation to gestational weight gain. Int J Environ Res Public Health 13:146.

Attanasio R 2019. Sex differences in the association between perfluoroalkyl acids and liver function in US adolescents: Analyses of NHANES 2013–2016. Environ Pollut 254:113061. [PubMed: 31454574]

Avanasi R, Shin HM, Vieira VM, Bartell SM. 2016a. Variability and epistemic uncertainty in water ingestion rates and pharmacokinetic parameters, and impact on the association between perfluorooctanoate and preeclampsia in the C8 Health Project population. Environ Res 146: 299–307. [PubMed: 26796985]

Avanasi R, Shin HM, Vieira VM, Savitz DA, Bartell SM. 2016b. Impact of exposure uncertainty on the association between perfluorooctanoate and preeclampsia in the C8 Health Project population. Environ Health Perspect 124:126–132. [PubMed: 26090912]

Bach CC, Liew Z, Bech BH, Nohr EA, Fei C, Bonefeld-Jorgensen EC, Henriksen TB, Olsen J. 2015. Perfluoroalkyl acids and time to pregnancy revisited: An update from the Danish National Birth Cohort. Environ Health 14:59. [PubMed: 26148742]

Ballesteros V, Costa O, Iniguez C, Fletcher T, Ballester F, Lopez-Espinosa MJ. 2017. Exposure to perfluoroalkyl substances and thyroid function in pregnant women and children: A systematic review of epidemiologic studies. Environ Int 99:15–28. [PubMed: 27884404]

Barry V, Darrow LA, Klein M, Winquist A, Steenland K. 2014. Early life perfluorooctanoic acid (PFOA) exposure and overweight and obesity risk in adulthood in a community with elevated exposure. Environ Res 132:62–69. [PubMed: 24742729]

Barry V, Winquist A, Steenland K. 2013. Perfluorooctanoic acid (PFOA) exposures and incident cancers among adults living near a chemical plant. Environ Health Perspect 121:1313–1318. [PubMed: 24007715]

Bartell SM, Calafat AM, Lyu C, Kato K, Ryan PB, Steenland K. 2010. Rate of decline in serum PFOA concentrations after granular activated carbon filtration at two public water systems in Ohio and West Virginia. Environ Health Perspect 118:222–228. [PubMed: 20123620]

Bassler J, Ducatman A, Elliott M, Wen S, Wahlang B, Barnett J, Cave MC. 2019. Environmental perfluoroalkyl acid exposures are associated with liver disease characterized by apoptosis and altered serum adipocytokines. Environ Pollut 247:1055–1063. [PubMed: 30823334]

Behr AC, Kwiatkowski A, Stahlman M, Schmidt FF, Luckert C, Braeuning A, Buhrke T. 2020a. Impairment of bile acid metabolism by perfluorooctanoic acid (PFOA) and perfluorooctanesulfonic acid (PFOS) in human HepaRG hepatoma cells. Arch Toxicol 94:1673–1686. [PubMed: 32253466]

Behr A-C, Lichtenstein D, Braeuning A, Lampen A, Buhrke T. 2018. Perfluoroalkylated substances (PFAS) affect neither estrogen and androgen receptor activity nor steroidogenesis in human cells in vitro. Toxicol Lett 291:51–60. [PubMed: 29601859]

Behr A-C, Plinsch C, Braeuning A, Buhrke T. 2020b. Activation of human nuclear receptors by perfluoroalkylated substances (PFAS). Toxicol In Vitro 62:104700. [PubMed: 31676336]

Benninghoff AD, Bisson WH, Koch DC, Ehresman DJ, Kolluri SK, Williams DE. 2010. Estrogen-like activity of perfluoroalkyl acids in vivo and interaction with human and rainbow trout estrogen receptors in vitro. Toxicol Sci 120:42–58. [PubMed: 21163906]

Berg V, Nost TH, Hansen S, Elverland A, Veyhe AS, Jorde R, Odland JO, Sandanger TM. 2015. Assessing the relationship between perfluoroalkyl substances, thyroid hormones and binding proteins in pregnant women; a longitudinal mixed effects approach. Environ Int 77:63–69. [PubMed: 25647630]

Bijland S, Rensen PC, Pieterman EJ, Maas AC, van der Hoorn JW, van Erk MJ, Havekes LM, Willems van Dijk K, Chang SC, Ehresman DJ, Butenhoff JL, Princen HMG. 2011. Perfluoroalkyl sulfonates cause alkyl chain length-dependent hepatic steatosis and hypolipidemia mainly by impairing lipoprotein production in APOE*3-Leiden CETP mice. Toxicol Sci 123:290–303. [PubMed: 21705711]

Bjerregaard-Olesen C, Bach CC, Long M, Wielsøe M, Bech BH, Henriksen TB, Olsen J, Bonefeld-Jørgensen EC. 2019. Associations of fetal growth outcomes with measures of the combined xenoestrogenic activity of maternal serum perfluorinated alkyl acids in Danish pregnant women. Environ Health Perspect 127:017006.

Bjork JA, Butenhoff JL, Wallace KB. 2011. Multiplicity of nuclear receptor activation by PFOA and PFOS in primary human and rodent hepatocytes. Toxicology 288:8–17. [PubMed: 21723365]

Blake BE, Cope HA, Hall SM, Keys RD, Mahler BW, McCord J, Scott B, Stapleton HM, Strynar MJ, Elmore SA, Fenton SE. 2020. Evaluation of maternal, embryo, and placental effects in CD-1 mice following gestational exposure to perfluorooctanoic acid (PFOA) or hexafluoropropylene oxide dimer acid (HFPO-DA or GenX). Environ Health Perspect 128:27006. [PubMed: 32074459]

Blake BE, Pinney SM, Hines EP, Fenton SE, Ferguson KK. 2018. Associations between longitudinal serum perfluoroalkyl substance (PFAS) levels and measures of thyroid hormone, kidney function, and body mass index in the Fernald Community Cohort. Environ Pollut 242:894–904. [PubMed: 30373035]

Bodin J, Groeng EC, Andreassen M, Dirven H, Nygaard UC. 2016. Exposure to perfluoroundecanoic acid (PFUnDA) accelerates insulitis development in a mouse model of type 1 diabetes. Toxicol Rep 3:664–672. [PubMed: 28959590]

Borghese MM, Walker M, Helewa ME, Fraser WD, Arbuckle TE. 2020. Association of perfluoroalkyl substances with gestational hypertension and preeclampsia in the MIREC study. Environ Int 141:105789. [PubMed: 32408216]

Borghoff SJ, Fitch S, Rager JE, Huggett D. 2018. A hypothesis-driven weight-of-evidence analysis to evaluate potential endocrine activity of perfluorohexanoic acid. Regul Toxicol Pharmacol 99:168–181. [PubMed: 30240830]

Bourre J-M, Francois M, Youyou A, Dumont O, Piciotti M, Pascal G, Durand G. 1989. The Effects of dietary α-linolenic acid on the composition of nerve membranes, enzymatic activity, amplitude of electrophysiological parameters, resistance to poisons and performance of learning tasks in rats. J Nutr 119:1880–1892. [PubMed: 2576038]

Buck RC, Franklin J, Berger U, Conder JM, Cousins IT, de Voogt P, Jensen AA, Kannan K, Mabury SA, van Leeuwen SP. 2011. Perfluoroalkyl and polyfluoroalkyl substances in the environment: Terminology, classification, and origins. Integr Environ Assess Manag 7:513–541. [PubMed: 21793199]

Buhrke T, Kruger E, Pevny S, Rossler M, Bitter K, Lampen A. 2015. Perfluorooctanoic acid (PFOA) affects distinct molecular signalling pathways in human primary hepatocytes. Toxicology 333:53–62. [PubMed: 25868421]

Butenhoff JL, Chang SC, Olsen GW, Thomford PJ. 2012. Chronic dietary toxicity and carcinogenicity study with potassium perfluorooctanesulfonate in Sprague Dawley rats. Toxicology 293:1–15. [PubMed: 22266392]

Butenhoff JL, Ehresman DJ, Chang SC, Parker GA, Stump DG. 2009. Gestational and lactational exposure to potassium perfluorooctanesulfonate (K+PFOS) in rats: Developmental neurotoxicity. Reprod Toxicol 27: 319–330. [PubMed: 19162172]

C8 Science Panel. 2012. Probable Link Evaluation of Thyroid disease. [cited 2020 July 13]. Available from: http://www.c8sciencepanel.org/pdfs/Probable_Link_C8_Thyroid_30Jul2012.pdf

Campbell S, Raza M, Pollack AZ. 2016. Perfluoroalkyl substances and endometriosis in US women in NHANES 2003–2006. Reprod Toxicol 65:230–235. [PubMed: 27544573]

Cardenas A, Gold DR, Hauser R, Kleinman KP, Hivert MF, Calafat AM, Ye X, Webster TF, Horton ES, Oken E. 2017. Plasma concentrations of per- and polyfluoroalkyl substances at baseline and associations with glycemic indicators and diabetes incidence among high-risk adults in the Diabetes Prevention Program Trial. Environ Health Perspect 125: 107001. [PubMed: 28974480]

Carr CK, Watkins AM, Wolf CJ, Abbott BD, Lau C, Gennings C. 2013. Testing for departures from additivity in mixtures of perfluoroalkyl acids (PFAAs). Toxicology 306:169–175. [PubMed: 23470359]

Casares D, Escribá PV, Rosselló CA. 2019. Membrane lipid composition: Effect on membrane and organelle structure, function and compartmentalization and therapeutic avenues. Int J Mol Sci 20:2167.

Caverly Rae JM, Craig L, Slone TW, Frame SR, Buxton LW, Kennedy GL. 2015. Evaluation of chronic toxicity and carcinogenicity of ammonium 2,3,3,3-tetrafluoro-2-(heptafluoropropoxy)-propanoate in Sprague-Dawley rats. Toxicol Rep 2:939–949. [PubMed: 28962433]

Centers for Disease Control and Prevention. 2017. Per- and polyfluorinated substances (PFAS) factsheet. [cited 2020 May 19]. Available from: https://www.cdc.gov/biomonitoring/PFAS_FactSheet.html

Chen J, Wang X, Ge X, Wang D, Wang T, Zhang L, Tanguay RL, Simonich M, Huang C, Dong Q. 2016. Chronic perfluorooctanesulphonic acid (PFOS) exposure produces estrogenic effects in zebrafish. Environ Pollut 218: 702–708. [PubMed: 27496563]

Chen P, Wang Q, Chen M, Yang J, Wang R, Zhong W, Zhu L, Yang L. 2018. Antagonistic estrogenic effects displayed by bisphenol AF and perfluorooctanoic acid on zebrafish (*Danio rerio*) at an early developmental stage. Environ Sci Technol Lett 5:655–661.

Cheng L, Albers P, Berney DM, Feldman DR, Daugaard G, Gilligan T, Looijenga LHJ. 2018. Testicular cancer. Nat Rev Dis Primers 4:29. [PubMed: 30291251]

Cheng W, Ng CA. 2017. A permeability-limited physiologically based pharmacokinetic (PBPK) model for perfluorooctanoic acid (PFOA) in male rats. Environ Sci Technol 51:9930–9939. [PubMed: 28759222]

Cheng W, Ng CA. 2018. Predicting relative protein affinity of novel per- and polyfluoroalkyl substances (PFASs) by an efficient molecular dynamics approach. Environ Sci Technol 52:7972–7980. [PubMed: 29897239]

Cheng W, Ng CA. 2019. Using machine learning to classify bioactivity for 3486 per- and polyfluoroalkyl substances (PFASs) from the OECD list. Environ Sci Technol 53:13970–13980. [PubMed: 31661253]

Chou WC, Lin Z. 2020. Probabilistic human health risk assessment of perfluorooctane sulfonate (PFOS) by integrating in vitro, in vivo toxicity, and human epidemiological studies using a Bayesian-based dose-response assessment coupled with physiologically based pharmacokinetic (PBPK) modeling approach. Environ Int 137:105581. [PubMed: 32087483]

Cluett R, Seshasayee SM, Rokoff LB, Rifas-Shiman SL, Ye X, Calafat AM, Gold DR, Coull B, Gordon CM, Rosen CJ, Oken E, Sagiv SK, Fleisch AF. 2019. Per- and polyfluoroalkyl substance plasma concentrations and bone mineral density in midchildhood: A cross-sectional study (Project Viva, United States). Environ Health Perspect 127:87006. [PubMed: 31433236]

Cohen DE, Fisher EA. 2013. Lipoprotein metabolism, dyslipidemia, and nonalcoholic fatty liver disease. Semin Liver Dis 33:380–388. [PubMed: 24222095]

Conley JM, Lambright CS, Evans N, Strynar MJ, McCord J, McIntyre BS, Travlos GS, Cardon MC, Medlock-Kakaley E, Hartig PC, Wilson VS, Gray LE Jr. 2019. Adverse maternal, fetal, and postnatal effects of hexafluoropropylene oxide dimer acid (GenX) from oral gestational exposure in Sprague-Dawley rats. Environ Health Perspect 127:37008. [PubMed: 30920876]

Convertino M, Church TR, Olsen GW, Liu Y, Doyle E, Elcombe CR, Barnett AL, Samuel LM, MacPherson IR, Evans TRJ. 2018. Stochastic pharmacokinetic-pharmacodynamic modeling for assessing the systemic health risk of perfluorooctanoate (PFOA). Toxicol Sci 163:293–306. [PubMed: 29462473]

Coperchini F, Awwad O, Rotondi M, Santini F, Imbriani M, Chiovato L. 2017. Thyroid disruption by perfluorooctane sulfonate (PFOS) and perfluorooctanoate (PFOA). J Endocrinol Invest 40:105–121. [PubMed: 27837466]

Corton JC, Peters JM, Klaunig JE. 2018. The PPARα-dependent rodent liver tumor response is not relevant to humans: Addressing misconceptions. Arch Toxicol 92:83–119. [PubMed: 29197930]

Cui L, Zhou QF, Liao CY, Fu JJ, Jiang GB. 2009. Studies on the toxicological effects of PFOA and PFOS on rats using histological observation and chemical analysis. Arch Environ Contam Toxicol 56:338–349. [PubMed: 18661093]

Daly ER, Chan BP, Talbot EA, Nassif J, Bean C, Cavallo SJ, Metcalf E, Simone K, Woolf AD. 2018. Per- and polyfluoroalkyl substance (PFAS) exposure assessment in a community exposed to contaminated drinking water, New Hampshire, 2015. Int J Hyg Environ Health 221: 569–577. [PubMed: 29514764]

Darrow LA, Groth AC, Winquist A, Shin HM, Bartell SM, Steenland K. 2016. Modeled perfluorooctanoic acid (PFOA) exposure and liver function in a mid-Ohio Valley community. Environ Health Perspect 124:1227–1233. [PubMed: 26978841]

Darrow LA, Stein CR, Steenland K. 2013. Serum perfluorooctanoic acid and perfluorooctane sulfonate concentrations in relation to birth outcomes in the mid-Ohio Valley, 2005–2010. Environ Health Perspect 121:1207–1213. [PubMed: 23838280]

Das KP, Grey BE, Rosen MB, Wood CR, Tatum-Gibbs KR, Zehr RD, Strynar MJ, Lindstrom AB, Lau C. 2015. Developmental toxicity of perfluorononanoic acid in mice. Reprod Toxicol 51:133–144. [PubMed: 25543169]

Das KP, Wood CR, Lin MT, Starkov AA, Lau C, Wallace KB, Corton JC, Abbott BD. 2017. Perfluoroalkyl acids-induced liver steatosis: Effects on genes controlling lipid homeostasis. Toxicology 378:37–52. [PubMed: 28049043]

DeWitt JC, Blossom SJ, Schaider LA. 2019. Exposure to per-fluoroalkyl and polyfluoroalkyl substances leads to immunotoxicity: Epidemiological and toxicological evidence. J Expo Sci Environ Epidemiol 29:148–156. [PubMed: 30482935]

Dhingra R, Winquist A, Darrow LA, Klein M, Steenland K. 2017. A study of reverse causation: Examining the associations of perfluorooctanoic acid serum levels with two outcomes. Environ Health Perspect 125:416–421. [PubMed: 27529882]

Di Nisio A, De Rocco Ponce M, Giadone A, Rocca MS, Guidolin D, Foresta C. 2020a. Perfluoroalkyl substances and bone health in young men: A pilot study. Endocrine 67:678–684. [PubMed: 31565782]

Di Nisio A, Rocca MS, Sabovic I, De Rocco Ponce M, Corsini C, Guidolin D, Zanon C, Acquasaliente L, Carosso AR, De Toni L, Foresta C. 2020b. Perfluorooctanoic acid alters progesterone activity in human endometrial cells and induces reproductive alterations in young women. Chemosphere 242:125208. [PubMed: 31896193]

Division of Science and Research, New Jersey Department of Environmental Protection. 2019. Interim specific ground water criterion for perfluorooctanoic acid (PFOA, C8) (CAS #: 335-67-1; chemical structure: CF3(CF2)6COOH)*. Trenton, NJ, USA. [cited 2020 July 13]. Available from: https://www.nj.gov/dep/dsr/supportdocs/PFOA_TSD.pdf

Dodes Traian MM, Cattoni DI, Levi V, Gonzalez Flecha FL. 2012. A two-stage model for lipid modulation of the activity of integral membrane proteins. PLoS One 7:e39255. [PubMed: 22723977]

Donat-Vargas C, Bergdahl IA, Tornevi A, Wennberg M, Sommar J, Kiviranta H, Koponen J, Rolandsson O, Akesson A. 2019. Perfluoroalkyl substances and risk of type II diabetes: A prospective nested case-control study. Environ Int 123:390–398. [PubMed: 30622063]

Dong Z, Wang H, Yu YY, Li YB, Naidu R, Liu Y. 2019. Using 2003–2014 U.S. NHANES data to determine the associations between per- and polyfluoroalkyl substances and cholesterol: Trend and implications. Ecotoxicol Environ Saf 173:461–468. [PubMed: 30798190]

EFSA Panel on Contaminants in the Food Chain. 2020. Public consultation on the draft scientific opinion on the risks to human health related to the presence of perfluoroalkyl substances in food. European Food Safety Authority, Parma, Italy. [cited 2020 September 15]. Available from: https://www.efsa.europa.eu/en/consultations/call/public-consultation-draft-scientific-opinion-risks-human-health

Elcombe CR, Elcombe BM, Foster JR, Chang SC, Ehresman DJ, Butenhoff JL. 2012. Hepatocellular hypertrophy and cell proliferation in Sprague-Dawley rats from dietary exposure to potassium perfluorooctanesulfonate results from increased expression of xenosensor nuclear receptors PPARα and CAR/PXR. Toxicology 293:16–29. [PubMed: 22245121]

Emmett EA, Shofer FS, Zhang H, Freeman D, Desai C, Shaw LM. 2006. Community exposure to perfluorooctanoate: Relationships between serum concentrations and exposure sources. J Occup Environ Med 48:759–770. [PubMed: 16902368]

Eriksen KT, Raaschou-Nielsen O, McLaughlin JK, Lipworth L, Tjonneland A, Overvad K, Sorensen M. 2013. Association between plasma PFOA and PFOS levels and total cholesterol in a middle-aged Danish population. PLoS One 8:e56969. [PubMed: 23441227]

Eryasa B, Grandjean P, Nielsen F, Valvi D, Zmirou-Navier D, Sunderland E, Weihe P, Oulhote Y. 2019. Physico-chemical properties and gestational diabetes predict transplacental transfer and partitioning of perfluoroalkyl substances. Environ Int 130:104874. [PubMed: 31200157]

European Environment Agency. 2019. Emerging chemical risks in Europe—"PFAS." Copenhagen, Denmark. [cited 2020 July 13]. Available from: https://www.eea.europa.eu/themes/human/chemicals/emerging-chemical-risks-in-europe

Fei C, McLaughlin JK, Lipworth L, Olsen J. 2009. Maternal levels of perfluorinated chemicals and subfecundity. Hum Reprod 24:1200–1205. [PubMed: 19176540]

Fenton SE, Reiner JL, Nakayama SF, Delinsky AD, Stanko JP, Hines EP, White SS, Lindstrom AB, Strynar MJ, Petropoulou SE. 2009. Analysis of PFOA in dosed CD-1 mice. Part 2. Disposition of PFOA in tissues and fluids from pregnant and lactating mice and their pups. Reprod Toxicol 27:365–372. [PubMed: 19429407]

Ferrari F, Orlando A, Ricci Z, Ronco C. 2019. Persistent pollutants: Focus on perfluorinated compounds and kidney. Curr Opin Crit Care 25:539–549. [PubMed: 31524719]

Filgo AJ, Quist EM, Hoenerhoff MJ, Brix AE, Kissling GE, Fenton SE. 2015. Perfluorooctanoic acid (PFOA)-induced liver lesions in two strains of mice following developmental exposures: PPARalpha is not required. Toxicol Pathol 43:558–568. [PubMed: 25398757]

Fisher M, Arbuckle TE, Wade M, Haines DA. 2013. Do perfluoroalkyl substances affect metabolic function and plasma lipids? Analysis of the 2007–2009, Canadian Health Measures Survey (CHMS) cycle 1. Environ Res 121:95–103. [PubMed: 23266098]

Fitz-Simon N, Fletcher T, Luster MI, Steenland K, Calafat AM, Kato K, Armstrong B. 2013. Reductions in serum lipids with a 4-year decline in serum perfluorooctanoic acid and perfluorooctanesulfonic acid. Epidemiology 24:569–576. [PubMed: 23685825]

Fletcher T, Galloway TS, Melzer D, Holcroft P, Cipelli R, Pilling LC, Mondal D, Luster M, Harries LW. 2013. Associations between PFOA, PFOS and changes in the expression of genes involved in cholesterol metabolism in humans. Environ Int 57–58:2–10.

Forhead AJ, Fowden AL. 2014. Thyroid hormones in fetal growth and prepartum maturation. J Endocrinol 221:R87–R103. [PubMed: 24648121]

Frisbee SJ, Shankar A, Knox SS, Steenland K, Savitz DA, Fletcher T, Ducatman AM. 2010. Perfluorooctanoic acid, perfluorooctanesulfonate, and serum lipids in children and adolescents: Results from the C8 Health Project. Arch Pediatr Adolesc Med 164:860–869. [PubMed: 20819969]

Fritsche K 2006. Fatty acids as modulators of the immune response. Annu Rev Nutr 26:45–73. [PubMed: 16848700]

Fromme H, Mosch C, Morovitz M, Alba-Alejandre I, Boehmer S, Kiranoglu M, Faber F, Hannibal I, Genzel-Boroviczeny O, Koletzko B, Volkel W. 2010. Pre- and postnatal exposure to perfluorinated compounds (PFCs). Environ Sci Technol 44:7123–7129. [PubMed: 20722423]

Fu Y, Wang T, Fu Q, Wang P, Lu Y. 2014. Associations between serum concentrations of perfluoroalkyl acids and serum lipid levels in a Chinese population. Ecotoxicol Environ Saf 106:246–252. [PubMed: 24863755]

Gaballah S, Swank A, Sobus JR, Howey XM, Schmid J, Catron T, McCord J, Hines E, Strynar M, Tal T. 2020. Evaluation of developmental toxicity, developmental neurotoxicity, and tissue dose in zebrafish exposed to GenX and other PFAS. Environ Health Perspect 128:47005. [PubMed: 32271623]

Gallo V, Leonardi G, Genser B, Lopez-Espinosa MJ, Frisbee SJ, Karlsson L, Ducatman AM, Fletcher T. 2012. Serum perfluorooctanoate (PFOA) and perfluorooctane sulfonate (PFOS) concentrations and liver function biomarkers in a population with elevated PFOA exposure. Environ Health Perspect 120:655–660. [PubMed: 22289616]

Gandaglia G, Ravi P, Abdollah F, Abd-El-Barr AE, Becker A, Popa I, Briganti A, Karakiewicz PI, Trinh QD, Jewett MA, Sun M. 2014. Contemporary incidence and mortality rates of kidney cancer in the United States. Can Urol Assoc J 8:247–252. [PubMed: 25210548]

Geiger SD, Xiao J, Shankar A. 2013. Positive association between perfluoroalkyl chemicals and hyperuricemia in children. Am J Epidemiol 177:1255–1262. [PubMed: 23552989]

Genser B, Teles CA, Barreto ML, Fischer JE. 2015. Within- and between-group regression for improving the robustness of causal claims in cross-sectional analysis. Environ Health 14:60. [PubMed: 26159541]

Girardi P, Merler E. 2019. A mortality study on male subjects exposed to polyfluoroalkyl acids with high internal dose of perfluorooctanoic acid. Environ Res 179:108743. [PubMed: 31542491]

Gleason JA, Post GB, Fagliano JA. 2015. Associations of perfluorinated chemical serum concentrations and biomarkers of liver function and uric acid in the US population (NHANES), 2007–2010. Environ Res 136:8–14. [PubMed: 25460614]

Gogola J, Hoffmann M, Ptak A. 2019. Persistent endocrine-disrupting chemicals found in human follicular fluid stimulate the proliferation of granulosa tumor spheroids via GPR30 and IGF1R but not via the classic estrogen receptors. Chemosphere 217:100–110. [PubMed: 30414542]

Goudarzi H, Miyashita C, Okada E, Kashino I, Chen CJ, Ito S, Araki A, Kobayashi S, Matsuura H, Kishi R. 2017. Prenatal exposure to perfluoroalkyl acids and prevalence of infectious diseases up to 4 years of age. Environ Int 104:132–138. [PubMed: 28392064]

Goulding DR, White SS, McBride SJ, Fenton SE, Harry GJ. 2017. Gestational exposure to perfluorooctanoic acid (PFOA): Alterations in motor related behaviors. Neurotoxicology 58:110–119. [PubMed: 27888120]

Graber JM, Alexander C, Laumbach RJ, Black K, Strickland PO, Georgopoulos PG, Marshall EG, Shendell DG, Alderson D, Mi Z, Mascari M, Weisel CP. 2019. Per- and polyfluoroalkyl substances (PFAS) blood levels after contamination of a community water supply and comparison with 2013–2014 NHANES. J Expo Sci Environ Epidemiol 29:172–182. [PubMed: 30482936]

Grandjean P, Andersen EW, Budtz-Jorgensen E, Nielsen F, Molbak K, Weihe P, Heilmann C. 2012. Serum vaccine antibody concentrations in children exposed to perfluorinated compounds. JAMA 307:391–397. [PubMed: 22274686]

Grandjean P, Heilmann C, Weihe P, Nielsen F, Mogensen UB, Budtz-Jorgensen E. 2017. Serum vaccine antibody concentrations in adolescents exposed to perfluorinated compounds. Environ Health Perspect 125:077018. [PubMed: 28749778]

Granum B, Haug LS, Namork E, Stolevik SB, Thomsen C, Aaberge IS, van Loveren H, Lovik M, Nygaard UC. 2013. Pre-natal exposure to perfluoroalkyl substances may be associated with altered vaccine antibody levels and immune-related health outcomes in early childhood. J Immunotoxicol 10:373–379. [PubMed: 23350954]

Guillette TC, McCord J, Guillette M, Polera ME, Rachels KT, Morgeson C, Kotlarz N, Knappe DRU, Reading BJ, Strynar M, Belcher SM. 2020. Elevated levels of per- and polyfluoroalkyl substances in Cape Fear River striped bass (*Morone saxatilis*) are associated with biomarkers of altered immune and liver function. Environ Int 136:105358. [PubMed: 32044175]

Guruge KS, Yeung LW, Yamanaka N, Miyazaki S, Lam PK, Giesy JP, Jones PD, Yamashita N. 2006. Gene expression profiles in rat liver treated with perfluorooctanoic acid (PFOA). Toxicol Sci 89:93–107. [PubMed: 16221955]

Gyllenhammar I, Benskin JP, Sandblom O, Berger U, Ahrens L, Lignell S, Wiberg K, Glynn A. 2018. Perfluoroalkyl acids (PFAAs) in serum from 2–4-month-old infants: Influence of maternal serum concentration, gestational age, breast-feeding, and contaminated drinking water. Environ Sci Technol 52:7101–7110. [PubMed: 29758986]

Hall JM, Greco CW. 2019. Perturbation of nuclear hormone receptors by endocrine disrupting chemicals: Mechanisms and pathological consequences of exposure. Cells 9:13.

Halldorsson TI, Rytter D, Haug LS, Bech BH, Danielsen I, Becher G, Henriksen TB, Olsen SF. 2012. Prenatal exposure to perfluorooctanoate and risk of overweight at 20 yr of age: A prospective cohort study. Environ Health Perspect 120:668–673. [PubMed: 22306490]

Han X, Nabb DL, Russell MH, Kennedy GL, Rickard RW. 2012. Renal elimination of perfluorocarboxylates (PFCAs). Chem Res Toxicol 25:35–46. [PubMed: 21985250]

He X, Liu Y, Xu B, Gu L, Tang W. 2018. PFOA is associated with diabetes and metabolic alteration in US men: National Health and Nutrition Examination Survey 2003–2012. Sci Total Environ 625:566–574. [PubMed: 29291571]

Hines EP, White SS, Stanko JP, Gibbs-Flournoy EA, Lau C, Fenton SE. 2009. Phenotypic dichotomy following developmental exposure to perfluorooctanoic acid (PFOA) in female CD-1 mice: Low doses induce elevated serum leptin and insulin, and overweight in mid-life. Mol Cell Endocrinol 304:97–105. [PubMed: 19433254]

Hoover G, Kar S, Guffey S, Leszczynski J, Sepulveda MS. 2019. In vitro and in silico modeling of perfluoroalkyl substances mixture toxicity in an amphibian fibroblast cell line. Chemosphere 233:25–33. [PubMed: 31163305]

Hu Y, Liu G, Rood J, Liang L, Bray GA, de Jonge L, Coull B, Furtado JD, Qi L, Grandjean P, Sun Q. 2019. Perfluoroalkyl substances and changes in bone mineral density: A prospective analysis in the POUNDS-LOST study. Environ Res 179:108775. [PubMed: 31593837]

Huang Q, Zhang J, Martin FL, Peng S, Tian M, Mu X, Shen H. 2013. Perfluorooctanoic acid induces apoptosis through the p53-dependent mitochondrial pathway in human hepatic cells: A proteomic study. Toxicol Lett 223:211–220. [PubMed: 24035753]

Huang R, Chen Q, Zhang L, Luo K, Chen L, Zhao S, Feng L, Zhang J. 2019. Prenatal exposure to perfluoroalkyl and polyfluoroalkyl substances and the risk of hypertensive disorders of pregnancy. Environ Health 18:5. [PubMed: 30626391]

Hui Z, Li R, Chen L. 2017. The impact of exposure to environmental contaminant on hepatocellular lipid metabolism. Gene 622:67–71. [PubMed: 28431976]

Huo X, Huang R, Gan Y, Luo K, Aimuzi R, Nian M, Ao J, Feng L, Tian Y, Wang W, Ye W, Zhang J, Shanghai Birth Cohort. 2020. Perfluoroalkyl substances in early pregnancy and risk of hypertensive disorders of pregnancy: A prospective cohort study. Environ Int 138:105656. [PubMed: 32222612]

IARC Working Group on the Evaluation of Carcinogenic Risks to Humans. 2017. Perfluorooctanoic acid. In Some Chemicals Used as Solvents and in Polymer Manufacture. IARC Monographs on the Evaluation of Carcinogenic Risks to Humans 110. International Agency for Research on Cancer, Lyon, France, pp 37–110.

Impinen A, Longnecker MP, Nygaard UC, London SJ, Ferguson KK, Haug LS, Granum B. 2019. Maternal levels of perfluoroalkyl substances (PFASs) during pregnancy and childhood allergy and asthma related outcomes and infections in the Norwegian Mother and Child (MoBa) cohort. Environ Int 124:462–472. [PubMed: 30684804]

Impinen A, Nygaard UC, Lodrup Carlsen KC, Mowinckel P, Carlsen KH, Haug LS, Granum B. 2018. Prenatal exposure to perfluoralkyl substances (PFASs) associated with respiratory tract infections but not allergy- and asthma-related health outcomes in childhood. Environ Res 160: 518–523. [PubMed: 29106950]

Inoue K, Ritz B, Andersen SL, Ramlau-Hansen CH, Hoyer BB, Bech BH, Henriksen TB, Bonefeld-Jorgensen EC, Olsen J, Liew Z. 2019. Perfluoroalkyl substances and maternal thyroid hormones in early pregnancy; findings in the Danish National Birth Cohort. Environ Health Perspect 127:117002. [PubMed: 31714155]

Interstate Technology and Regulatory Council. 2020. Chemistry, terminology and acronyms. [cited 2020 May 19]. Available from: https://pfas-1.itrcweb.org/2-2-chemistry-terminology-and-acronyms/

Jain RB. 2020. Variabilities in concentrations of selected perfluoroalkyl acids among normotensives and hypertensives across various stages of glomerular function. Arch Environ Occup Health 2020:1–11.

Jain RB, Ducatman A. 2019a. Dynamics of associations between perfluoroalkyl substances and uric acid across the various stages of glomerular function. Environ Sci Pollut Res Int 26:12425–12434. [PubMed: 30847808]

Jain RB, Ducatman A. 2019b. Perfluoroalkyl acids serum concentrations and their relationship to biomarkers of renal failure: Serum and urine albumin, creatinine, and albumin creatinine ratios across the spectrum of glomerular function among US adults. Environ Res 174:143–151. [PubMed: 31077989]

Jain RB, Ducatman A. 2019c. Perfluoroalkyl substances follow inverted U-shaped distributions across various stages of glomerular function: Implications for future research. Environ Res 169:476–482. [PubMed: 30530087]

Jain RB, Ducatman A. 2019d. Roles of gender and obesity in defining correlations between perfluoroalkyl substances and lipid/lipoproteins. Sci Total Environ 653:74–81. [PubMed: 30408670]

Jain RB, Ducatman A. 2019e. Selective associations of recent low concentrations of perfluoroalkyl substances with liver function biomarkers: NHANES 2011 to 2014 data on US adults aged ≥20 years. J Occup Environ Med 61:293–302. [PubMed: 30589657]

Jensen T, Niwa K, Hisatome I, Kanbay M, Andres-Hernando A, Roncal-Jimenez CA, Sato Y, Garcia G, Ohno M, Lanaspa MA, Johnson RJ, Kuwabara M. 2018. Increased serum uric acid over five years is a risk factor for developing fatty liver. Sci Rep 8:11735. [PubMed: 30082907]

Jian JM, Chen D, Han FJ, Guo Y, Zeng L, Lu X, Wang F. 2018. A short review on human exposure to and tissue distribution of per- and polyfluoroalkyl substances (PFASs). Sci Total Environ 636:1058–1069. [PubMed: 29913568]

Jin R, McConnell R, Catherine C, Xu S, Walker DI, Stratakis N, Jones DP, Miller GW, Peng C, Conti DV, Vos MB, Chatzi L. 2020. Perfluoroalkyl substances and severity of nonalcoholic fatty liver in children: An untargeted metabolomics approach. Environ Int 134:105220. [PubMed: 31744629]

Joensen UN, Bossi R, Leffers H, Jensen AA, Skakkebaek NE, Jorgensen N. 2009. Do perfluoroalkyl compounds impair human semen quality? Environ Health Perspect 117:923–927. [PubMed: 19590684]

Johnson MS, Buck RC, Cousins IT, Weis CP, Fenton SE. 2020. Estimating environmental hazard and risks from exposure to per- and polyfluoroalkyl substances (PFAS): Outcome of a SETAC focused topic meeting. Environ Toxicol Chem, in press. 10.1002/etc.4784

Johnson PI, Sutton P, Atchley DS, Koustas E, Lam J, Sen S, Robinson KA, Axelrad DA, Woodruff TJ. 2014. The Navigation Guide—Evidence-based medicine meets environmental health: Systematic review of human evidence for PFOA effects on fetal growth. Environ Health Perspect 122:1028–1039. [PubMed: 24968388]

Jones PD, Hu W, De Coen W, Newsted JL, Giesy JP. 2003. Binding of perfluorinated fatty acids to serum proteins. Environ Toxicol Chem 22:2639–2649. [PubMed: 14587903]

Kang Q, Gao F, Zhang X, Wang L, Liu J, Fu M, Zhang S, Wan Y, Shen H, Hu J. 2020. Nontargeted identification of per- and polyfluoroalkyl substances in human follicular fluid and their blood-follicle transfer. Environ Int 139:105686. [PubMed: 32278886]

Karnes C, Winquist A, Steenland K. 2014. Incidence of type II diabetes in a cohort with substantial exposure to perfluorooctanoic acid. Environ Res 128:78–83. [PubMed: 24299613]

Katakura M, Kudo N, Tsuda T, Hibino Y, Mitsumoto A, Kawashima Y. 2007. Rat organic anion transporter 3 and organic anion transporting polypeptide 1 mediate perfluorooctanoic acid transport. J Health Sci 53:77–83.

Kataria A, Trachtman H, Malaga-Dieguez L, Trasande L. 2015. Association between perfluoroalkyl acids and kidney function in a cross-sectional study of adolescents. Environ Health 14:89. [PubMed: 26590127]

Kato K, Wong LY, Jia LT, Kuklenyik Z, Calafat AM. 2011. Trends in exposure to polyfluoroalkyl chemicals in the U.S. population: 1999–2008. Environ Sci Technol 45:8037–8045. [PubMed: 21469664]

Khalil N, Chen A, Lee M, Czerwinski SA, Ebert JR, DeWitt JC, Kannan K. 2016. Association of perfluoroalkyl substances, bone mineral density, and osteoporosis in the U.S. population in NHANES 2009–2010. Environ Health Perspect 124:81–87. [PubMed: 26058082]

Kilkenny C, Browne WJ, Cuthill IC, Emerson M, Altman DG. 2010. Improving bioscience research reporting: The ARRIVE guidelines for reporting animal research. PLoS Biol 8:e1000412. [PubMed: 20613859]

Kim DH, Kim UJ, Kim HY, Choi SD, Oh JE. 2016. Perfluoroalkyl substances in serum from South Korean infants with congenital hypothyroidism and healthy infants—Its relationship with thyroid hormones. Environ Res 147:399–404. [PubMed: 26950028]

Kingsley SL, Walker DI, Calafat AM, Chen A, Papandonatos GD, Xu Y, Jones DP, Lanphear BP, Pennell KD, Braun JM. 2019. Metabolomics of childhood exposure to perfluoroalkyl substances: A cross-sectional study. Metabolomics 15:95. [PubMed: 31227916]

Kjeldsen LS, Bonefeld-Jørgensen EC. 2013. Perfluorinated compounds affect the function of sex hormone receptors. Environ Sci Pollut Res 20:8031–8044.

Klaunig JE, Hocevar BA, Kamendulis LM. 2012. Mode of action analysis of perfluorooctanoic acid (PFOA) tumorigenicity and human relevance. Reprod Toxicol 33:410–418. [PubMed: 22120428]

Koponen J, Winkens K, Airaksinen R, Berger U, Vestergren R, Cousins IT, Karvonen AM, Pekkanen J, Kiviranta H. 2018. Longitudinal trends of per- and polyfluoroalkyl substances in children's serum. Environ Int 121:591–599. [PubMed: 30308470]

Koshy TT, Attina TM, Ghassabian A, Gilbert J, Burdine LK, Marmor M, Honda M, Chu DB, Han X, Shao Y, Kannan K, Urbina EM, Trasande L. 2017. Serum perfluoroalkyl substances and cardiometabolic consequences in adolescents exposed to the World Trade Center disaster and a matched comparison group. Environ Int 109:128–135. [PubMed: 28890218]

Koustas E, Lam J, Sutton P, Johnson PI, Atchley DS, Sen S, Robinson KA, Axelrad DA, Woodruff TJ. 2014. The Navigation Guide—Evidence-based medicine meets environmental health: Systematic review of nonhuman evidence for PFOA effects on fetal growth. Environ Health Perspect 122:1015–1027. [PubMed: 24968374]

Kvalem HE, Nygaard UC, Lodrup Carlsen KC, Carlsen KH, Haug LS, Granum B. 2020. Perfluoroalkyl substances, airways infections, allergy and asthma related health outcomes—Implications of gender, exposure period and study design. Environ Int 134:105259. [PubMed: 31733527]

LaLone CA, Ankley GT, Belanger SE, Embry MR, Hodges G, Knapen D, Munn S, Perkins EJ, Rudd MA, Villeneuve DL, Whelan M, Willett C, Zhang X, Hecker M. 2017. Advancing the adverse outcome pathway framework—An international horizon scanning approach. Environ Toxicol Chem 36:1411–1421. [PubMed: 28543973]

Lam J, Koustas E, Sutton P, Johnson PI, Atchley DS, Sen S, Robinson KA, Axelrad DA, Woodruff TJ. 2014. The Navigation Guide—Evidence-based medicine meets environmental health: Integration of animal and human evidence for PFOA effects on fetal growth. Environ Health Perspect 122:1040–1051. [PubMed: 24968389]

Lau C, Anitole K, Hodes C, Lai D, Pfahles-Hutchens A, Seed J. 2007. Perfluoroalkyl acids: A review of monitoring and toxicological findings. Toxicol Sci 99:366–394. [PubMed: 17519394]

Lau C, Thibodeaux JR, Hanson RG, Narotsky MG, Rogers JM, Lindstrom AB, Strynar MJ. 2006. Effects of perfluorooctanoic acid exposure during pregnancy in the mouse. Toxicol Sci 90:510–518. [PubMed: 16415327]

Lee JE, Choi K. 2017. Perfluoroalkyl substances exposure and thyroid hormones in humans: Epidemiological observations and implications. Ann Pediatr Endocrinol Metab 22:6–14. [PubMed: 28443254]

Li C-H, Ren X-M, Cao L-Y, Qin W-P, Guo L-H. 2019. Investigation of binding and activity of perfluoroalkyl substances to the human peroxisome proliferator-activated receptor β/δ. Environ Sci Process Impacts 21: 1908–1914. [PubMed: 31332417]

Li K, Sun J, Yang J, Roberts SM, Zhang X, Cui X, Wei S, Ma LQ. 2017a. Molecular mechanisms of perfluorooctanoate-induced hepatocyte apoptosis in mice using proteomic techniques. Environ Sci Technol 51:11380–11389. [PubMed: 28885018]

Li Y, Barregard L, Xu Y, Scott K, Pineda D, Lindh CH, Jakobsson K, Fletcher T. 2020. Associations between perfluoroalkyl substances and serum lipids in a Swedish adult population with contaminated drinking water. Environ Health 19:33. [PubMed: 32169067]

Li Y, Cheng Y, Xie Z, Zeng F. 2017b. Perfluorinated alkyl substances in serum of the southern Chinese general population and potential impact on thyroid hormones. Sci Rep 7:43380. [PubMed: 28240244]

Li Y, Fletcher T, Mucs D, Scott K, Lindh CH, Tallving P, Jakobsson K. 2018. Half-lives of PFOS, PFHxS and PFOA after end of exposure to contaminated drinking water. Occup Environ Med 75:46–51. [PubMed: 29133598]

Liberatore HK, Jackson SR, Strynar MJ, McCord JP. 2020. Solvent suitability for HFPO-DA (“GenX” parent acid) in toxicological studies. Environ Sci Technol Lett 7:477–481. [PubMed: 32944590]

Liew Z, Goudarzi H, Oulhote Y. 2018. Developmental exposures to perfluoroalkyl substances (PFASs): An update of associated health outcomes. Curr Environ Health Rep 5:1–19. [PubMed: 29556975]

Lin CY, Lin LY, Chiang CK, Wang WJ, Su YN, Hung KY, Chen PC. 2010. Investigation of the associations between low-dose serum perfluorinated chemicals and liver enzymes in US adults. Am J Gastroenterol 105: 1354–1363. [PubMed: 20010922]

Lin LY, Wen LL, Su TC, Chen PC, Lin CY. 2014. Negative association between serum perfluorooctane sulfate concentration and bone mineral density in US premenopausal women: NHANES, 2005–2008. J Clin Endocrinol Metab 99:2173–2180. [PubMed: 24606077]

Lin PD, Cardenas A, Hauser R, Gold DR, Kleinman KP, Hivert MF, Fleisch AF, Calafat AM, Webster TF, Horton ES, Oken E. 2019. Per- and polyfluoroalkyl substances and blood lipid levels in pre-diabetic adults-longitudinal analysis of the diabetes prevention program outcomes study. Environ Int 129:343–353. [PubMed: 31150976]

Liu G, Zhang B, Hu Y, Rood J, Liang L, Qi L, Bray GA, DeJonge L, Coull B, Grandjean P, Furtado JD, Sun Q. 2020. Associations of perfluoroalkyl substances with blood lipids and apolipoproteins in lipoprotein sub-species: The POUNDS-lost study. Environ Health 19:5. [PubMed: 31931806]

Liu S, Yang R, Yin N, Faiola F. 2020. The short-chain perfluorinated compounds PFBS, PFHxS, PFBA and PFHxA, disrupt human mesenchymal stem cell self-renewal and adipogenic differentiation. J Environ Sci 88:187–199.

Looker C, Luster MI, Calafat AM, Johnson VJ, Burleson GR, Burleson FG, Fletcher T. 2014. Influenza vaccine response in adults exposed to perfluorooctanoate and perfluorooctanesulfonate. Toxicol Sci 138:76–88. [PubMed: 24284791]

Lopez-Espinosa MJ, Mondal D, Armstrong B, Bloom MS, Fletcher T. 2012. Thyroid function and perfluoroalkyl acids in children living near a chemical plant. Environ Health Perspect 120:1036–1041. [PubMed: 22453676]

Louis GM, Chen Z, Schisterman EF, Kim S, Sweeney AM, Sundaram R, Lynch CD, Gore-Langton RE, Barr DB. 2015. Perfluorochemicals and human semen quality: The LIFE study. Environ Health Perspect 123:57–63. [PubMed: 25127343]

Louis GM, Peterson CM, Chen Z, Hediger ML, Croughan MS, Sundaram R, Stanford JB, Fujimoto VY, Varner MW, Giudice LC, Kennedy A, Sun L, Wu Q, Kannan K. 2012. Perfluorochemicals and endometriosis: The ENDO study. Epidemiology 23:799–805. [PubMed: 22992575]

Luebker DJ, Hansen KJ, Bass NM, Butenhoff JL, Seacat AM. 2002. Interactions of fluorochemicals with rat liver fatty acid-binding protein. Toxicology 176:175–185. [PubMed: 12093614]

Lum KJ, Sundaram R, Barr DB, Louis TA, Buck, Louis GM. 2017. Perfluoroalkyl chemicals, menstrual cycle length, and fecundity: Findings from a prospective pregnancy study. Epidemiology 28:90–98. [PubMed: 27541842]

Lundin JI, Alexander BH, Olsen GW, Church TR. 2009. Ammonium perfluorooctanoate production and occupational mortality. Epidemiology 20:921–928. [PubMed: 19797969]

Macon MB, Villanueva LR, Tatum-Gibbs K, Zehr RD, Strynar MJ, Stanko JP, White SS, Helfant L, Fenton SE. 2011. Prenatal perfluorooctanoic acid exposure in CD-1 mice: Low-dose developmental effects and internal dosimetry. Toxicol Sci 122:134–145. [PubMed: 21482639]

Maestri L, Negri S, Ferrari M, Ghittori S, Fabris F, Danesino P, Imbriani M. 2006. Determination of perfluorooctanoic acid and perfluorooctanesulfonate in human tissues by liquid chromatography/single quadrupole mass spectrometry. Rapid Commun Mass Spectrom 20:2728–2734. [PubMed: 16915561]

Martin MT, Brennan RJ, Hu W, Ayanoglu E, Lau C, Ren H, Wood CR, Corton JC, Kavlock RJ, Dix DJ. 2007. Toxicogenomic study of triazole fungicides and perfluoroalkyl acids in rat livers predicts toxicity and categorizes chemicals based on mechanisms of toxicity. Toxicol Sci 97: 595–613. [PubMed: 17383973]

Martinsson M, Nielsen C, Bjork J, Rylander L, Malmqvist E, Lindh C, Rignell-Hydbom A. 2020. Intrauterine exposure to perfluorinated compounds and overweight at age 4: A case-control study. PLoS One 15:e0230137. [PubMed: 32176721]

Massoud O, Charlton M. 2018. Nonalcoholic fatty liver disease/nonalcoholic steatohepatitis and hepatocellular carcinoma. Clin Liver Dis 22:201–211. [PubMed: 29128057]

Mastrantonio M, Bai E, Uccelli R, Cordiano V, Screpanti A, Crosignani P. 2018. Drinking water contamination from perfluoroalkyl substances (PFAS): An ecological mortality study in the Veneto region, Italy. Eur J Public Health 28:180–185. [PubMed: 28541558]

McCord J, Strynar M. 2019. Identification of per- and polyfluoroalkyl substances in the Cape Fear River by high resolution mass spectrometry and nontargeted screening. Environ Sci Technol 53:4717–4727. [PubMed: 30993978]

Meek ME, Boobis A, Cote I, Dellarco V, Fotakis G, Munn S, Seed J, Vickers C. 2014. New developments in the evolution and application of the WHO/IPCS framework on mode of action/species concordance analysis. J Appl Toxicol 34:1–18. [PubMed: 24166207]

Melzer D, Rice N, Depledge MH, Henley WE, Galloway TS. 2010. Association between serum perfluorooctanoic acid (PFOA) and thyroid disease in the U.S. National Health and Nutrition Examination Survey. Environ Health Perspect 118:686–692. [PubMed: 20089479]

Michigan PFAS Science Advisory Panel. 2018. Scientific evidence and recommendations for managing PFAS contamination in Michigan. Lansing, MI, USA. [cited 2020 July 13]. Available from: https://www.michigan.gov/documents/pfasresponse/Science_Advisory_Board_Report_641294_7.pdf

Nakagawa H, Hirata T, Terada T, Jutabha P, Miura D, Harada KH, Inoue K, Anzai N, Endou H, Inui K, Kanai Y, Koizumi A. 2008. Roles of organic anion transporters in the renal excretion of perfluorooctanoic acid. Basic Clin Pharmacol Toxicol 103:1–8. [PubMed: 18373647]

National Toxicology Program. 2016. Immunotoxicity associated with exposure to perfluorooctanoic acid (PFOA) or perfluorooctane sulfonate (PFOS). US Department of Health and Human Services, Research Triangle Park, NC. [cited 2020 July 13]. Available from: https://ntp.niehs.nih.gov/ntp/ohat/pfoa_pfos/pfoa_pfosmonograph_508.pdf

National Toxicology Program. 2020a. NTP technical report on the toxicology and carcinogenesis studies of perfluorooctanoic acid (CAS no. 335-67-1) administered in feed to Sprague Dawley (Hsd:Sprague Dawley® SD®) rats. Technical Report 598. US Department of Health and Human Services, Research Triangle Park, NC. [cited 2020 September 15]. Available from: https://ntp.niehs.nih.gov/ntp/about_ntp/trpanel/2019/december/tr598draft.pdf

National Toxicology Program. 2020b. P08: Statistical analysis of primary tumors—Perfluorooctanoic acid. US Department of Health and Human Services, Research Triangle Park, NC. [cited 2020 September 15]. Available from: https://www.documentcloud.org/documents/6155302-Statistical-Analysis-Tumors.html

National Toxicology Program. 2020c. Testing status of perfluorooctanoic acid (PFOA) M910070. US Department of Health and Human Services, Research Triangle Park, NC. [cited 2020 May 19]. Available from: https://ntp.niehs.nih.gov/go/ts-m910070

Nelson JW, Hatch EE, Webster TF. 2010. Exposure to polyfluoroalkyl chemicals and cholesterol, body weight, and insulin resistance in the general U.S. population. Environ Health Perspect 118:197–202. [PubMed: 20123614]

New Jersey Drinking Water Quality Institute Health Effects Subcommittee. 2017. Health-based maximum contaminant level support document: Perfluorooctanoic acid (PFOA). Trenton, NJ, USA. [cited 2020 July 13]. Available from: https://www.state.nj.us/dep/watersupply/pdf/pfoa-appendixa.pdf

Ng CA, Hungerbühler K. 2014. Bioaccumulation of perfluorinated alkyl acids: Observations and models. Environ Sci Technol 48:4637–4648. [PubMed: 24762048]

Ng CA, Hungerbuehler K. 2015. Exploring the use of molecular docking to identify bioaccumulative perfluorinated alkyl acids (PFAAs). Environ Sci Technol 49:12306–12314. [PubMed: 26393377]

Ngo HT, Hetland RB, Sabaredzovic A, Haug LS, Steffensen IL. 2014. In utero exposure to perfluorooctanoate (PFOA) or perfluorooctane sulfonate (PFOS) did not increase body weight or intestinal tumorigenesis in multiple intestinal neoplasia (Min/+) mice. Environ Res 132:251–263. [PubMed: 24834819]

Nian M, Li QQ, Bloom M, Qian ZM, Syberg KM, Vaughn MG, Wang SQ, Wei Q, Zeeshan M, Gurram N, Chu C, Wang J, Tian Y-P, Hu L-W, Liu K-K, Yang B-Y, Liu R-Q, Feng D, Zeng X-W, Dong G-H. 2019. Liver function biomarkers disorder is associated with exposure to perfluoroalkyl acids in adults: Isomers of C8 Health Project in China. Environ Res 172:81–88. [PubMed: 30776734]

Obermayr RP, Temml C, Gutjahr G, Knechtelsdorfer M, Oberbauer R, Klauser-Braun R. 2008. Elevated uric acid increases the risk for kidney disease. J Am Soc Nephrol 19:2407–2413. [PubMed: 18799720]

Olsen GW, Burris JM, Ehresman DJ, Froehlich JW, Seacat AM, Butenhoff JL, Zobel LR. 2007. Half-life of serum elimination of perfluorooctanesulfonate, perfluorohexanesulfonate, and perfluorooctanoate in retired fluorochemical production workers. Environ Health Perspect 115:1298–1305. [PubMed: 17805419]

Olsen GW, Chang SC, Noker PE, Gorman GS, Ehresman DJ, Lieder PH, Butenhoff JL. 2009. A comparison of the pharmacokinetics of perfluorobutanesulfonate (PFBS) in rats, monkeys, and humans. Toxicology 256:65–74. [PubMed: 19059455]

Olsen GW, Mair DC, Lange CC, Harrington LM, Church TR, Goldberg CL, Herron RM, Hanna H, Nobiletti JB, Rios JA, Reagan WK, Ley CA. 2017. Per- and polyfluoroalkyl substances (PFAS) in American Red Cross adult blood donors, 2000–2015. Environ Res 157:87–95. [PubMed: 28528142]

Organisation for Economic Co-operation Development. 2017. Revised guidance document on developing and assessing adverse outcome pathways. Series on Testing and Assessment, No. 184. ENV/JM/MONO (2013)6 Paris, France. [cited 2020 July 13]. Available from: http://www.oecd.org/officialdocuments/publicdisplaydocumentpdf/?cote=env/jm/mono(2013)6&doclanguage=en

Organisation for Economic Co-operation Development. 2018. Toward a new comprehensive global database of per-and polyfluoroalkyl substances (PFASs): Summary report on updating the OECD 2007 list of per-and polyfluoroalkyl substances (PFASs). ENV/JM/MONO(2018)7. Series on Risk Management, No. 39 Paris, France. [cited 2020 July 13]. Available from: https://www.oecd.org/officialdocuments/publicdisplaydocumentpdf/?cote=ENV-JM-MONO(2018)7&doclanguage=en

Organisation for Economic Co-operation Development. 2020. AOPs. [cited 2020 May 19]. Available from: https://aopwiki.org/aops

Pachkowski B, Post GB, Stern AH. 2019. The derivation of a reference dose (RfD) for perfluorooctane sulfonate (PFOS) based on immune suppression. Environ Res 171:452–469. [PubMed: 30739020]

Pan Y, Cui Q, Wang J, Sheng N, Jing J, Yao B, Dai J. 2019. Profiles of emerging and legacy per-/polyfluoroalkyl substances in matched serum and semen samples: New implications for human semen quality. Environ Health Perspect 127:127005. [PubMed: 31841032]

Papadopoulou E, Sabaredzovic A, Namork E, Nygaard UC, Granum B, Haug LS. 2016. Exposure of Norwegian toddlers to perfluoroalkyl substances (PFAS): The association with breastfeeding and maternal PFAS concentrations. Environ Int 94:687–694. [PubMed: 27453094]

Park JS, Kim J, Elghiaty A, Ham WS. 2018. Recent global trends in testicular cancer incidence and mortality. Medicine 97:e12390. [PubMed: 30213007]

Patlewicz G, Richard AM, Williams AJ, Grulke CM, Sams R, Lambert J, Noyes PD, DeVito MJ, Hines RN, Strynar M, Guiseppe-Elie A, Thomas RS. 2019. A chemical category-based prioritization approach for selecting 75 per- and polyfluoroalkyl substances (PFAS) for tiered toxicity and toxicokinetic testing. Environ Health Perspect 127: 014501.

Perez F, Nadal M, Navarro-Ortega A, Fabrega F, Domingo JL, Barcelo D, Farre M. 2013. Accumulation of perfluoroalkyl substances in human tissues. Environ Int 59:354–362. [PubMed: 23892228]

Perla FM, Prelati M, Lavorato M, Visicchio D, Anania C. 2017. The role of lipid and lipoprotein metabolism in non-alcoholic fatty liver disease. Children (Basel) 4:46.

Post GB. 2020. Recent US state and federal drinking water guidelines for per- and polyfluoroalkyl substances (PFAS). Environ Toxicol Chem, in press. 10.1002/etc.4863

Post GB, Gleason JA, Cooper KR. 2017. Key scientific issues in developing drinking water guidelines for perfluoroalkyl acids: Contaminants of emerging concern. PLoS Biol 15:e2002855. [PubMed: 29261653]

Pouwer MG, Pieterman EJ, Chang SC, Olsen GW, Caspers MPM, Verschuren L, Jukema JW, Princen HMG. 2019. Dose effects of ammonium perfluorooctanoate on lipoprotein metabolism in APOE*3-Leiden.CETP mice. Toxicol Sci 168:519–534. [PubMed: 30657992]

Qin WP, Cao LY, Li CH, Guo LH, Colbourne J, Ren XM. 2020. Perfluoroalkyl substances stimulate insulin secretion by islet beta cells via G protein-coupled receptor 40. Environ Sci Technol 54:3428–3436. [PubMed: 32092270]

Qin XD, Qian Z, Vaughn MG, Huang J, Ward P, Zeng XW, Zhou Y, Zhu Y, Yuan P, Li M, Bai Z, Paul G, Hao Y-T, Chen W, Chen P-C, Dong G-H, Lee YL. 2016. Positive associations of serum perfluoroalkyl substances with uric acid and hyperuricemia in children from Taiwan. Environ Pollut 212:519–524. [PubMed: 26970855]

Qin XD, Qian ZM, Dharmage SC, Perret J, Geiger SD, Rigdon SE, Howard S, Zeng XW, Hu LW, Yang BY, Zhou Y, Li M, Xu S-L, Bao W-W, Zhang Y-Z, Yuan P, Wang J, Zhang C, Tian Y-P, Nian M, Xiao X, Chen W, Lee YL, Dong G-H. 2017. Association of perfluoroalkyl substances exposure with impaired lung function in children. Environ Res 155:15–21. [PubMed: 28171771]

Qiu Z, Qu K, Luan F, Liu Y, Zhu Y, Yuan Y, Li H, Zhang H, Hai Y, Zhao C. 2020. Binding specificities of estrogen receptor with perfluorinated compounds: A cross species comparison. Environ Int 134:105284. [PubMed: 31707300]

Quist EM, Filgo AJ, Cummings CA, Kissling GE, Hoenerhoff MJ, Fenton SE. 2015. Hepatic mitochondrial alteration in CD-1 mice associated with prenatal exposures to low doses of perfluorooctanoic acid (PFOA). Toxicol Pathol 43:546–557. [PubMed: 25326589]

Rabinowitz JR, Goldsmith M-R, Little SB, Pasquinelli MA. 2008. Computational molecular modeling for evaluating the toxicity of environmental chemicals: Prioritizing bioassay requirements. Environ Health Perspect 116:573–577. [PubMed: 18470285]

Rantakokko P, Mannisto V, Airaksinen R, Koponen J, Viluksela M, Kiviranta H, Pihlajamaki J. 2015. Persistent organic pollutants and non-alcoholic fatty liver disease in morbidly obese patients: A cohort study. Environ Health 14:79. [PubMed: 26420011]

Rappazzo KM, Coffman E, Hines EP. 2017. Exposure to perfluorinated alkyl substances and health outcomes in children: A systematic review of the epidemiologic literature. Int J Environ Res Public Health 14:691.

Rashid F, Ramakrishnan A, Fields C, Irudayaraj J. 2020. Acute PFOA exposure promotes epigenomic alterations in mouse kidney tissues. Toxicol Rep 7:125–132. [PubMed: 31938689]

Rebholz SL, Jones T, Herrick RL, Xie C, Calafat AM, Pinney SM, Woollett LA. 2016. Hypercholesterolemia with consumption of PFOA-laced Western diets is dependent on strain and sex of mice. Toxicol Rep 3:46–54. [PubMed: 26942110]

Ren XM, Qin WP, Cao LY, Zhang J, Yang Y, Wan B, Guo LH. 2016. Binding interactions of perfluoroalkyl substances with thyroid hormone transport proteins and potential toxicological implications. Toxicology 366–367: 32–42.

Romano ME, Xu Y, Calafat AM, Yolton K, Chen A, Webster GM, Eliot MN, Howard CR, Lanphear BP, Braun JM. 2016. Maternal serum perfluoroalkyl substances during pregnancy and duration of breastfeeding. Environ Res 149:239–246. [PubMed: 27179585]

Rosen EM, Brantsaeter AL, Carroll R, Haug L, Singer AB, Zhao S, Ferguson KK. 2018. Maternal plasma concentrations of per- and polyfluoroalkyl substances and breastfeeding duration in the Norwegian Mother and Child Cohort. Environ Epidemiol 2:e027. [PubMed: 30298140]

Rosen MB, Das KP, Rooney J, Abbott B, Lau C, Corton JC. 2017. PPARalpha-independent transcriptional targets of perfluoroalkyl acids revealed by transcript profiling. Toxicology 387:95–107. [PubMed: 28558994]

Russell MH, Nilsson H, Buck RC. 2013. Elimination kinetics of perfluorohexanoic acid in humans and comparison with mouse, rat and monkey. Chemosphere 93:2419–2425. [PubMed: 24050716]

Sabovic I, Cosci I, De Toni L, Ferramosca A, Stornaiuolo M, Di Nisio A, Dall'Acqua S, Garolla A, Foresta C. 2020. Perfluoro-octanoic acid impairs sperm motility through the alteration of plasma membrane. J Endocrinol Invest 43:641–652. [PubMed: 31776969]

Sakr CJ, Kreckmann KH, Green JW, Gillies PJ, Reynolds JL, Leonard RC. 2007a. Cross-sectional study of lipids and liver enzymes related to a serum biomarker of exposure (ammonium perfluorooctanoate or APFO) as part of a general health survey in a cohort of occupationally exposed workers. J Occup Environ Med 49:1086–1096. [PubMed: 18000414]

Sakr CJ, Leonard RC, Kreckmann KH, Slade MD, Cullen MR. 2007b. Longitudinal study of serum lipids and liver enzymes in workers with occupational exposure to ammonium perfluorooctanoate. J Occup Environ Med 49:872–879. [PubMed: 17693785]

Sakuma A, Wasada Ochi H, Yoshioka M, Yamanaka N, Ikezawa M, Guruge KS. 2019. Changes in hepato-renal gene expression in microminipigs following a single exposure to a mixture of perfluoroalkyl acids. PLoS One 14:e0210110. [PubMed: 30608957]

Salihovic S, Dickens AM, Schoultz I, Fart F, Sinisalu L, Lindeman T, Halfvarson J, Oresic M, Hyotylainen T. 2020. Simultaneous determination of perfluoroalkyl substances and bile acids in human serum using ultra-high-performance liquid chromatography-tandem mass spectrometry. Anal Bioanal Chem 412:2251–2259. [PubMed: 31760452]

Salihovic S, Fall T, Ganna A, Broeckling CD, Prenni JE, Hyotylainen T, Karrman A, Lind PM, Ingelsson E, Lind L. 2019. Identification of metabolic profiles associated with human exposure to perfluoroalkyl substances. J Expo Sci Environ Epidemiol 29:196–205. [PubMed: 30185940]

Salvalaglio M, Muscionico I, Cavallotti C. 2010. Determination of energies and sites of binding of PFOA and PFOS to human serum albumin. J Phys Chem B 114:14860–14874. [PubMed: 21028884]

Sattar N, Forrest E, Preiss D. 2014. Non-alcoholic fatty liver disease. BMJ 349:g4596. [PubMed: 25239614]

Savitz DA, Stein CR, Bartell SM, Elston B, Gong J, Shin HM, Wellenius GA. 2012. Perfluorooctanoic acid exposure and pregnancy outcome in a highly exposed community. Epidemiology 23:386–392. [PubMed: 22370857]

Schlezinger J, Puckett H, Oliver J, Nielsen G, Heiger-Bernays W, Webster T. 2020. Perfluorooctanoic acid activates multiple nuclear receptor pathways and skews expression of genes regulating cholesterol homeostasis in liver of humanized PPARα mice fed an American diet. Toxicol Appl Pharmacol 405:115204. [PubMed: 32822737]

Seo SH, Son MH, Choi SD, Lee DH, Chang YS. 2018. Influence of exposure to perfluoroalkyl substances (PFASs) on the Korean general population: 10-year trend and health effects. Environ Int 113:149–161. [PubMed: 29425899]

Sha B, Schymanski EL, Ruttkies C, Cousins IT, Wang Z. 2019. Exploring open cheminformatics approaches for categorizing per- and polyfluoroalkyl substances (PFASs). Environ Sci Process Impacts 21:1835–1851. [PubMed: 31576380]

Shabalina IG, Kalinovich AV, Cannon B, Nedergaard J. 2016. Metabolically inert perfluorinated fatty acids directly activate uncoupling protein 1 in brown-fat mitochondria. Arch Toxicol 90:1117–1128. [PubMed: 26041126]

Shankar A, Xiao J, Ducatman A. 2011. Perfluoroalkyl chemicals and chronic kidney disease in US adults. Am J Epidemiol 174:893–900. [PubMed: 21873601]

Sheng N, Cui R, Wang J, Guo Y, Wang J, Dai J. 2018. Cytotoxicity of novel fluorinated alternatives to long-chain perfluoroalkyl substances to human liver cell line and their binding capacity to human liver fatty acid binding protein. Arch Toxicol 92:359–369. [PubMed: 28864880]

Skuladottir M, Ramel A, Rytter D, Haug LS, Sabaredzovic A, Bech BH, Henriksen TB, Olsen SF, Halldorsson TI. 2015. Examining confounding by diet in the association between perfluoroalkyl acids and serum cholesterol in pregnancy. Environ Res 143:33–38. [PubMed: 26432473]

Slotkin TA, MacKillop EA, Melnick RL, Thayer KA, Seidler FJ. 2008. Developmental neurotoxicity of perfluorinated chemicals modeled in vitro. Environ Health Perspect 116:716–722. [PubMed: 18560525]

Solo-Gabriele HM, Jones AS, Lindstrom AB, Lang JR. 2020. Waste type, incineration, and aeration are associated with per- and polyfluoroalkyl levels in landfill leachates. Waste Manag 107:191–200. [PubMed: 32304853]

Song M, Kim YJ, Park YK, Ryu JC. 2012. Changes in thyroid peroxidase activity in response to various chemicals. J Environ Monit 14:2121–2126. [PubMed: 22699773]

Song X, Tang S, Zhu H, Chen Z, Zang Z, Zhang Y, Niu X, Wang X, Yin H, Zeng F, He C. 2018. Biomonitoring PFAAs in blood and semen samples: Investigation of a potential link between PFAAs exposure and semen mobility in China. Environ Int 113:50–54. [PubMed: 29421407]

Spector AA, Yorek MA. 1985. Membrane lipid composition and cellular function. J Lipid Res 26:1015–1035. [PubMed: 3906008]

Stanifer JW, Stapleton HM, Souma T, Wittmer A, Zhao X, Boulware LE. 2018. Perfluorinated chemicals as emerging environmental threats to kidney health: A scoping review. Clin J Am Soc Nephrol 13:1479–1492. [PubMed: 30213782]

Starling AP, Engel SM, Whitworth KW, Richardson DB, Stuebe AM, Daniels JL, Haug LS, Eggesbo M, Becher G, Sabaredzovic A, Eggesbo M, Hoppin JA, Travlos GS, Wilson RE, Trogstad LI, Magnus P, Longnecker MP. 2014. Perfluoroalkyl substances and lipid concentrations in plasma during pregnancy among women in the Norwegian Mother and Child Cohort Study. Environ Int 62:104–112. [PubMed: 24189199]

Steenland K, Barry V, Savitz D. 2018a. Serum perfluorooctanoic acid and birthweight: An updated meta-analysis with bias analysis. Epidemiology 29:765–776. [PubMed: 30063543]

Steenland K, Kugathasan S, Barr DB. 2018b. PFOA and ulcerative colitis. Environ Res 165:317–321. [PubMed: 29777922]

Steenland K, Tinker S, Frisbee S, Ducatman A, Vaccarino V. 2009. Association of perfluorooctanoic acid and perfluorooctane sulfonate with serum lipids among adults living near a chemical plant. Am J Epidemiol 170:1268–1278. [PubMed: 19846564]

Steenland K, Tinker S, Shankar A, Ducatman A. 2010. Association of perfluorooctanoic acid (PFOA) and perfluorooctane sulfonate (PFOS) with uric acid among adults with elevated community exposure to PFOA. Environ Health Perspect 118:229–233. [PubMed: 20123605]

Steenland K, Woskie S. 2012. Cohort mortality study of workers exposed to perfluorooctanoic acid. Am J Epidemiol 176:909–917. [PubMed: 23079607]

Steenland K, Zhao L, Winquist A. 2015. A cohort incidence study of workers exposed to perfluorooctanoic acid (PFOA). Occup Environ Med 72:373–380. [PubMed: 25601914]

Steenland K, Zhao L, Winquist A, Parks C. 2013. Ulcerative colitis and perfluorooctanoic acid (PFOA) in a highly exposed population of community residents and workers in the mid-Ohio Valley. Environ Health Perspect 121:900–905. [PubMed: 23735465]

Stein CR, McGovern KJ, Pajak AM, Maglione PJ, Wolff MS. 2016. Perfluoroalkyl and polyfluoroalkyl substances and indicators of immune function in children aged 12–19 y: National Health and Nutrition Examination Survey. Pediatr Res 79:348–357. [PubMed: 26492286]

Stein CR, Savitz DA, Dougan M. 2009. Serum levels of perfluorooctanoic acid and perfluorooctane sulfonate and pregnancy outcome. Am J Epidemiol 170:837–846. [PubMed: 19692329]

Stubleski J, Salihovic S, Lind L, Lind PM, van Bavel B, Karrman A. 2016. Changes in serum levels of perfluoroalkyl substances during a 10-year follow-up period in a large population-based cohort. Environ Int 95:86–92. [PubMed: 27542758]

Sun Q, Zong G, Valvi D, Nielsen F, Coull B, Grandjean P. 2018. Plasma concentrations of perfluoroalkyl substances and risk of type 2 diabetes: A prospective investigation among U.S. women. Environ Health Perspect 126:037001. [PubMed: 29498927]

Sunderland EM, Hu XC, Dassuncao C, Tokranov AK, Wagner CC, Allen JG. 2019. A review of the pathways of human exposure to poly- and perfluoroalkyl substances (PFASs) and present understanding of health effects. J Expo Sci Environ Epidemiol 29:131–147. [PubMed: 30470793]

Susmann HP, Schaider LA, Rodgers KM, Rudel RA. 2019. Dietary habits related to food packaging and population exposure to PFASs. Environ Health Perspect 127:107003. [PubMed: 31596611]

Szilagyi JT, Freedman AN, Kepper SL, Keshava AM, Bangma JT, Fry RC. 2020. Per- and polyfluoroalkyl substances (PFAS) differentially inhibit placental trophoblast migration and invasion in vitro. Toxicol Sci 175:210–219. [PubMed: 32219433]

Tadic M, Cuspidi C, Vasic D, Kerkhof PLM. 2018. Cardiovascular implications of diabetes, metabolic syndrome, thyroid disease, and cardiooncology in women. In Kerkhof PLM, Miller VM, eds, Sex-Specific Analysis of Cardiovascular Function. Springer International, Cham, Switzerland, pp 471–488.

Tan X, Xie G, Sun X, Li Q, Zhong W, Qiao P, Sun X, Jia W, Zhou Z. 2013. High fat diet feeding exaggerates perfluorooctanoic acid-induced liver injury in mice via modulating multiple metabolic pathways. PLoS One 8:e61409. [PubMed: 23626681]

Temkin AM, Hocevar BA, Andrews DQ, Naidenko OV, Kamendulis LM. 2020. Application of the key characteristics of carcinogens to per and polyfluoroalkyl substances. Int J Environ Res Public Health 17:1668.

Thibodeaux JR, Hanson RG, Rogers JM, Grey BE, Barbee BD, Richards JH, Butenhoff JL, Stevenson LA, Lau C. 2003. Exposure to perfluorooctane sulfonate during pregnancy in rat and mouse. I: Maternal and prenatal evaluations. Toxicol Sci 74:369–381. [PubMed: 12773773]

Tilton SC, Orner GA, Benninghoff AD, Carpenter HM, Hendricks JD, Pereira CB, Williams DE. 2008. Genomic profiling reveals an alternate mechanism for hepatic tumor promotion by perfluorooctanoic acid in rainbow trout. Environ Health Perspect 116:1047–1055. [PubMed: 18709148]

Timmermann CA, Budtz-Jorgensen E, Jensen TK, Osuna CE, Petersen MS, Steuerwald U, Nielsen F, Poulsen LK, Weihe P, Grandjean P. 2017a. Association between perfluoroalkyl substance exposure and asthma and allergic disease in children as modified by MMR vaccination. J Immunotoxicol 14:39–49. [PubMed: 28091126]

Timmermann CA, Rossing LI, Grontved A, Ried-Larsen M, Dalgard C, Andersen LB, Grandjean P, Nielsen F, Svendsen KD, Scheike T, Jensen TK. 2014. Adiposity and glycemic control in children exposed to perfluorinated compounds. J Clin Endocrinol Metab 99:E608–E614. [PubMed: 24606078]

Timmermann CAG, Budtz-Jorgensen E, Petersen MS, Weihe P, Steuerwald U, Nielsen F, Jensen TK, Grandjean P. 2017b. Shorter duration of breastfeeding at elevated exposures to perfluoroalkyl substances. Reprod Toxicol 68:164–170. [PubMed: 27421579]

Tucker DK, Macon MB, Strynar MJ, Dagnino S, Andersen E, Fenton SE. 2015. The mammary gland is a sensitive pubertal target in CD-1 and C57Bl/6 mice following perinatal perfluorooctanoic acid (PFOA) exposure. Reprod Toxicol 54:26–36. [PubMed: 25499722]

US Environmental Protection Agency. 2018. PFAS structures in DSSTox. Washington, DC. [cited 2020 May 19]. Available from: https://comptox.epa.gov/dashboard/chemical_lists/PFASSTRUCT

US Environmental Protection Agency. 2019. EPA’s per- and polyfluoroalkyl substances (PFAS) action plan. EPA 823R18004. Washington, DC. [cited 2020 July 13]. Available from: www.epa.gov/pfas

US Environmental Protection Agency. 2020. Announcement of preliminary regulatory determinations for contaminants on the fourth drinking water contaminant candidate list. 85 FR 14098 Washington, DC. [cited 2020 September 15]. Available from: https://www.federalregister.gov/documents/2020/03/10/2020-04145/announcement-of-preliminary-regulatory-determinations-for-contaminants-on-the-fourth-drinking-water

van Esterik JCJ, Sales LB, Dollé MET, Håkansson H, Herlin M, Legler J, van der Ven LTM. 2016. Programming of metabolic effects in C57BL/6JxFVB mice by in utero and lactational exposure to perfluorooctanoic acid. Arch Toxicol 90:701–715. [PubMed: 25827101]

VanNoy BN, Lam J, Zota AR. 2018. Breastfeeding as a predictor of serum concentrations of per- and polyfluorinated alkyl substances in reproductive-aged women and young children: A rapid systematic review. Curr Environ Health Rep 5:213–224. [PubMed: 29737463]

Velez MP, Arbuckle TE, Fraser WD. 2015. Maternal exposure to perfluorinated chemicals and reduced fecundity: The MIREC study. Hum Reprod 30:701–709. [PubMed: 25567616]

Vested A, Ramlau-Hansen CH, Olsen SF, Bonde JP, Kristensen SL, Halldorsson TI, Becher G, Haug LS, Ernst EH, Toft G. 2013. Associations of in utero exposure to perfluorinated alkyl acids with human semen quality and reproductive hormones in adult men. Environ Health Perspect 121:453–458. [PubMed: 23360585]

Viberg H, Lee I, Eriksson P. 2013. Adult dose-dependent behavioral and cognitive disturbances after a single neonatal PFHxS dose. Toxicology 304:185–191. [PubMed: 23287389]

Vieira VM, Hoffman K, Shin HM, Weinberg JM, Webster TF, Fletcher T. 2013. Perfluorooctanoic acid exposure and cancer outcomes in a contaminated community: A geographic analysis. Environ Health Perspect 121:318–323. [PubMed: 23308854]

Wahlang B, Jin J, Beier JI, Hardesty JE, Daly EF, Schnegelberger RD, Falkner KC, Prough RA, Kirpich IA, Cave MC. 2019. Mechanisms of environmental contributions to fatty liver disease. Curr Environ Health Rep 6:80–94. [PubMed: 31134516]

Wan HT, Zhao YG, Wei X, Hui KY, Giesy JP, Wong CK. 2012. PFOS-induced hepatic steatosis, the mechanistic actions on beta-oxidation and lipid transport. Biochim Biophys Acta 1820:1092–1101. [PubMed: 22484034]

Wang B, Zhang R, Jin F, Lou H, Mao Y, Zhu W, Zhou W, Zhang P, Zhang J. 2017. Perfluoroalkyl substances and endometriosis-related infertility in Chinese women. Environ Int 102:207–212. [PubMed: 28283302]

Wang L, Wang Y, Liang Y, Li J, Liu Y, Zhang J, Zhang A, Fu J, Jiang G. 2013. Specific accumulation of lipid droplets in hepatocyte nuclei of PFOA-exposed BALB/c mice. Sci Rep 3:2174. [PubMed: 23846197]

Wang L, Wang Y, Liang Y, Li J, Liu Y, Zhang J, Zhang A, Fu J, Jiang G. 2014. PFOS induced lipid metabolism disturbances in BALB/c mice through inhibition of low density lipoproteins excretion. Sci Rep 4:4582. [PubMed: 24694979]

Wang X, Bai Y, Tang C, Cao X, Chang F, Chen L. 2018. Impact of perfluorooctane sulfonate on reproductive ability of female mice through suppression of estrogen receptor alpha-activated kisspeptin neurons. Toxicol Sci 165:475–486. [PubMed: 29939337]

Wang Z, DeWitt JC, Higgins CP, Cousins IT. 2017. A never-ending story of per- and polyfluoroalkyl substances (PFASs)? Environ Sci Technol 51: 2508–2518. [PubMed: 28224793]

Waterfield G, Rogers M, Grandjean P, Auffhammer M, Sunding D. 2020. Reducing exposure to high levels of perfluorinated compounds in drinking water improves reproductive outcomes: Evidence from an intervention in Minnesota. Environ Health 19:42. [PubMed: 32321520]

Watkins DJ, Josson J, Elston B, Bartell SM, Shin HM, Vieira VM, Savitz DA, Fletcher T, Wellenius GA. 2013. Exposure to perfluoroalkyl acids and markers of kidney function among children and adolescents living near a chemical plant. Environ Health Perspect 121:625–630. [PubMed: 23482063]

Weaver YM, Ehresman DJ, Butenhoff JL, Hagenbuch B. 2009. Roles of rat renal organic anion transporters in transporting perfluorinated carboxylates with different chain lengths. Toxicol Sci 113:305–314. [PubMed: 19915082]

Webster GM, Rauch SA, Marie NS, Mattman A, Lanphear BP, Venners SA. 2016. Cross-sectional associations of serum perfluoroalkyl acids and thyroid hormones in U.S. adults: Variation according to TPOAb and iodine status (NHANES 2007–2008). Environ Health Perspect 124:935–942. [PubMed: 26517287]

Webster GM, Venners SA, Mattman A, Martin JW. 2014. Associations between perfluoroalkyl acids (PFASs) and maternal thyroid hormones in early pregnancy: A population-based cohort study. Environ Res 133: 338–347. [PubMed: 25019470]

Wei Y, Dai J, Liu M, Wang J, Xu M, Zha J, Wang Z. 2009. Estrogen-like properties of perfluorooctanoic acid as revealed by expressing hepatic estrogen-responsive genes in rare minnows (*Gobiocypris rarus*). Environ Toxicol Chem 26:2440–2447.

Wen LL, Lin CY, Chou HC, Chang CC, Lo HY, Juan SH. 2016. Perfluorooctanesulfonate mediates renal tubular cell apoptosis through PPARgamma inactivation. PLoS One 11:e0155190. [PubMed: 27171144]

White SS, Calafat AM, Kuklenyik Z, Villanueva L, Zehr RD, Helfant L, Strynar MJ, Lindstrom AB, Thibodeaux JR, Wood C, Fenton SE. 2007. Gestational PFOA exposure of mice is associated with altered mammary gland development in dams and female offspring. Toxicol Sci 96:133–144. [PubMed: 17132714]

White SS, Fenton SE, Hines EP. 2011a. Endocrine disrupting properties of perfluorooctanoic acid. J Steroid Biochem Mol Biol 127:16–26. [PubMed: 21397692]

White SS, Stanko JP, Kato K, Calafat AM, Hines EP, Fenton SE. 2011b. Gestational and chronic low-dose PFOA exposures and mammary gland growth and differentiation in three generations of CD-1 mice. Environ Health Perspect 119:1070–1076. [PubMed: 21501981]

Whitworth KW, Haug LS, Baird DD, Becher G, Hoppin JA, Skjaerven R, Thomsen C, Eggesbo M, Travlos G, Wilson R, Longnecker MP. 2012. Perfluorinated compounds and subfecundity in pregnant women. Epidemiology 23:257–263. [PubMed: 22081060]

Wikstrom S, Lin PI, Lindh CH, Shu H, Bornehag CG. 2020. Maternal serum levels of perfluoroalkyl substances in early pregnancy and offspring birth weight. Pediatr Res 87:1093–1099. [PubMed: 31835271]

Wikstrom S, Lindh CH, Shu H, Bornehag CG. 2019. Early pregnancy serum levels of perfluoroalkyl substances and risk of preeclampsia in Swedish women. Sci Rep 9:9179. [PubMed: 31235847]

Williams AJ, Grulke CM, Edwards J, McEachran AD, Mansouri K, Baker NC, Patlewicz G, Shah I, Wambaugh JF, Judson RS, Richard AM. 2017. The CompTox Chemistry Dashboard: A community data resource for environmental chemistry. J Cheminform 9:61. [PubMed: 29185060]

Winquist A, Steenland K. 2014a. Modeled PFOA exposure and coronary artery disease, hypertension, and high cholesterol in community and worker cohorts. Environ Health Perspect 122:1299–1305. [PubMed: 25260175]

Winquist A, Steenland K. 2014b. Perfluorooctanoic acid exposure and thyroid disease in community and worker cohorts. Epidemiology 25:255–264. [PubMed: 24407430]

Wolf CJ, Rider CV, Lau C, Abbott BD. 2014. Evaluating the additivity of perfluoroalkyl acids in binary combinations on peroxisome proliferator-activated receptor-alpha activation. Toxicology 316:43–54. [PubMed: 24374136]

Wolf CJ, Schmid JE, Lau C, Abbott BD. 2012. Activation of mouse and human peroxisome proliferator-activated receptor-alpha (PPARα) by perfluoroalkyl acids (PFAAs): Further investigation of C4–C12 compounds. Reprod Toxicol 33:546–551. [PubMed: 22107727]

Wolf CJ, Takacs ML, Schmid JE, Lau C, Abbott BD. 2008. Activation of mouse and human peroxisome proliferator-activated receptor alpha by perfluoroalkyl acids of different functional groups and chain lengths. Toxicol Sci 106:162–171. [PubMed: 18713766]

World Health Organization. 2020. Mode of action framework (for cancer and non-cancer risk assessment). Geneva, Switzerland. [cited 2020 May 19]. Available from: https://www.who.int/ipcs/methods/harmonization/areas/cancer/en/

Xiao C, Grandjean P, Valvi D, Nielsen F, Jensen TK, Weihe P, Oulhote Y. 2020. Associations of exposure to perfluoroalkyl substances with thyroid hormone concentrations and birth size. J Clin Endocrinol Metab 105:735–745.

Xu HE, Lambert MH, Montana VG, Parks DJ, Blanchard SG, Brown PJ, Sternbach DD, Lehmann JM, Wisely GB, Willson TM, Kliewer SA, Milburn MV. 1999. Molecular recognition of fatty acids by peroxisome proliferator-activated receptors. Mol Cell 3:397–403. [PubMed: 10198642]

Xu J, Shimpi P, Armstrong L, Salter D, Slitt AL. 2016. PFOS induces adipogenesis and glucose uptake in association with activation of Nrf2 signaling pathway. Toxicol Appl Pharmacol 290:21–30. [PubMed: 26548598]

Xu M, Liu G, Li M, Huo M, Zong W, Liu R. 2020a. Probing the cell apoptosis pathway induced by perfluorooctanoic acid and perfluorooctane sulfonate at the subcellular and molecular levels. J Agric Food Chem 68: 633–641. [PubMed: 31891488]

Xu Y, Li Y, Scott K, Lindh CH, Jakobsson K, Fletcher T, Ohlsson B, Andersson EM. 2020b. Inflammatory bowel disease and biomarkers of gut inflammation and permeability in a community with high exposure to perfluoroalkyl substances through drinking water. Environ Res 181:108923. [PubMed: 31759646]

Yamaguchi M, Arisawa K, Uemura H, Katsuura-Kamano S, Takami H, Sawachika F, Nakamoto M, Juta T, Toda E, Mori K, Hasegawa M, Tanto M, Shima M, Sumiyoshi Y, Morinaga K, Kodama K, Suzuki T, Nagai M, Satoh H. 2013. Consumption of seafood, serum liver enzymes, and blood levels of PFOS and PFOA in the Japanese population. J Occup Health 55:184–194. [PubMed: 23574777]

Yang CH, Glover KP, Han X. 2009. Organic anion transporting polypeptide (Oatp) 1a1-mediated perfluorooctanoate transport and evidence for a renal reabsorption mechanism of Oatp1a1 in renal elimination of perfluorocarboxylates in rats. Toxicol Lett 190:163–171. [PubMed: 19616083]

Yao X, Sha S, Wang Y, Sun X, Cao J, Kang J, Jiang L, Chen M, Ma Y. 2016. Perfluorooctane sulfonate induces autophagy-dependent apoptosis through spinster 1-mediated lysosomal-mitochondrial axis and impaired mitophagy. Toxicol Sci 153:198–211. [PubMed: 27413105]

Yuan Y, Ding X, Cheng Y, Kang H, Luo T, Zhang X, Kuang H, Chen Y, Zeng X, Zhang D. 2020. PFOA evokes extracellular Ca^{2+} influx and compromises progesterone-induced response in human sperm. Chemosphere 241: 125074. [PubMed: 31627108]

Zeilmaker M, Fragki S, Verbruggen E, Bokkers B, Lijzen J. 2018. Mixture exposure to PFAS: A relative potency factor approach. RIVM-2018–0070. Rijksinstituut voor Volksgezondheid en Milieu, Bilthoven, Netherlands. [cited 2020 July 13]. Available from: 10.21945/rivm-2018-0070

Zeng XW, Lodge CJ, Dharmage SC, Bloom MS, Yu Y, Yang M, Chu C, Li QQ, Hu LW, Liu KK, Yang B-Y, Dong G-H. 2019. Isomers of per- and polyfluoroalkyl substances and uric acid in adults: Isomers of C8 Health Project in China. Environ Int 133:105160. [PubMed: 31518937]

Zeng XW, Qian Z, Emo B, Vaughn M, Bao J, Qin XD, Zhu Y, Li J, Lee YL, Dong GH. 2015. Association of polyfluoroalkyl chemical exposure with serum lipids in children. Sci Total Environ 512–513:364–370.

Zhang H, He J, Li N, Gao N, Du Q, Chen B, Chen F, Shan X, Ding Y, Zhu W, Wu Y, Tang J, Jia X. 2019. Lipid accumulation responses in the liver of *Rana nigromaculata* induced by perfluorooctanoic acid (PFOA). Ecotoxicol Environ Saf 167:29–35. [PubMed: 30292973]

Zhang J, Begum A, Brannstrom K, Grundstrom C, Iakovleva I, Olofsson A, Sauer-Eriksson AE, Andersson PL. 2016. Structure-based virtual screening protocol for in silico identification of potential thyroid disrupting chemicals targeting transthyretin. Environ Sci Technol 50:11984–11993. [PubMed: 27668830]

Zhang L, Duan X, Sun W, Sun H. 2020. Perfluorooctane sulfonate acute exposure stimulates insulin secretion via GPR40 pathway. Sci Total Environ 726:138498. [PubMed: 32305757]

Zhang L, Krishnan P, Ehresman DJ, Smith PB, Dutta M, Bagley BD, Chang SC, Butenhoff JL, Patterson AD, Peters JM. 2016. Editor's highlight: Perfluorooctane sulfonate-choline ion pair formation: A potential mechanism modulating hepatic steatosis and oxidative stress in mice. Toxicol Sci 153:186–197. [PubMed: 27413108]

Zhang Y, Beesoon S, Zhu L, Martin JW. 2013. Biomonitoring of perfluoroalkyl acids in human urine and estimates of biological half-life. Environ Sci Technol 47:10619–10627. [PubMed: 23980546]

Zhang Y, Cao X, Chen L, Qin Y, Xu Y, Tian Y, Chen L. 2020. Exposure of female mice to perfluorooctanoic acid suppresses hypothalamic kisspeptin-reproductive endocrine system through enhanced hepatic fibroblast growth factor 21 synthesis, leading to ovulation failure and prolonged dioestrus. J Neuroendocrinol 32:e12848. [PubMed: 32307816]

Zhao J, Hinton P, Chen J, Jiang J. 2020. Causal inference for the effect of environmental chemicals on chronic kidney disease. Comput Struct Biotechnol J 18:93–99. [PubMed: 31934310]

Zhou R, Cheng W, Feng Y, Wei H, Liang F, Wang Y. 2017. Interactions between three typical endocrine-disrupting chemicals (EDCs) in binary mixtures exposure on myocardial differentiation of mouse embryonic stem cell. Chemosphere 178:378–383. [PubMed: 28340460]

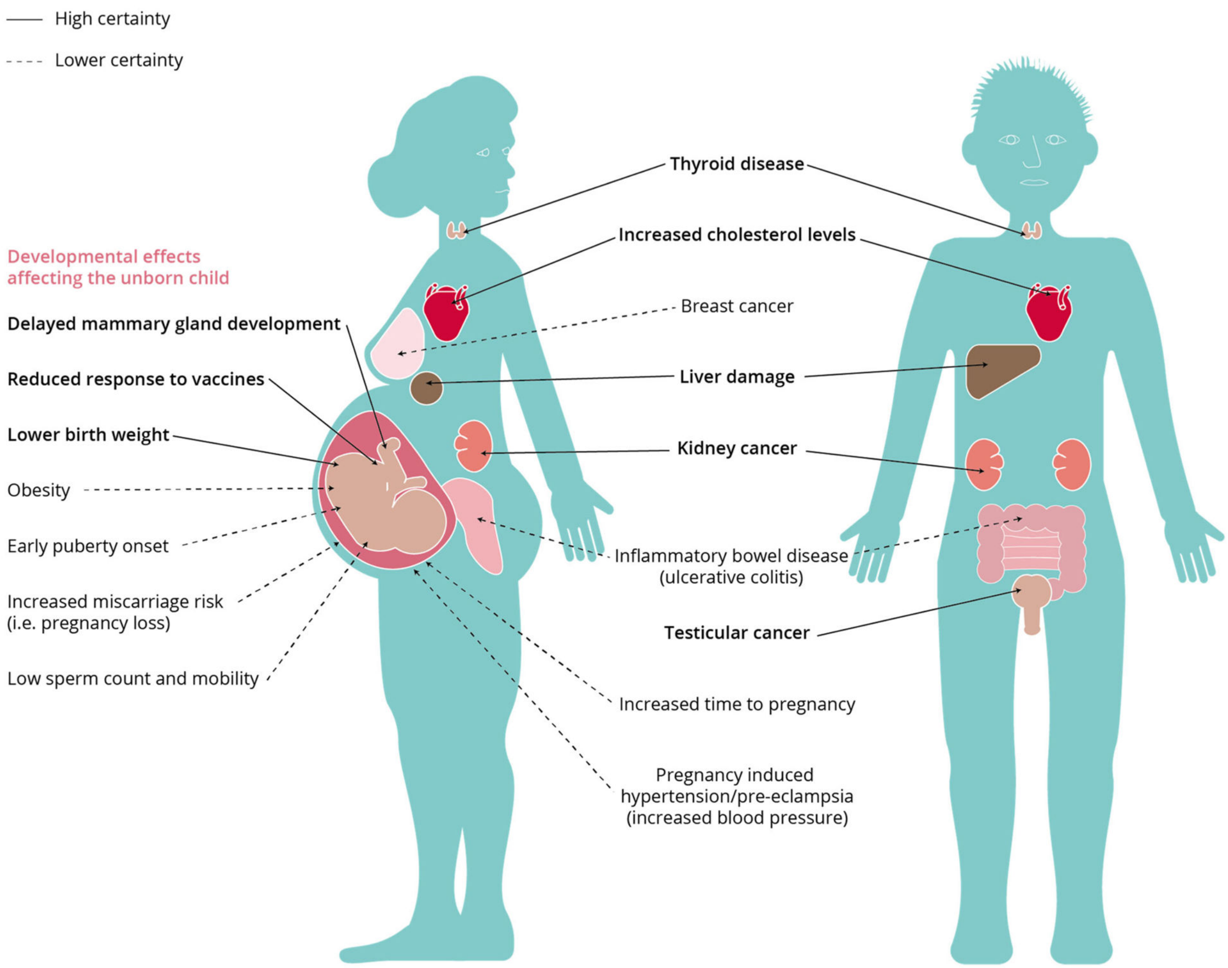

FIGURE 1:

Effects of per- and polyfluoroalkyl substances on human health. Used with permission from European Environment Agency (2019). Original sources for this figure: National Toxicology Program (2016), C8 Science Panel (2012), IARC Working Group on the Evaluation of Carcinogenic Risks to Humans (2017), Barry et al. (2013), Fenton et al. (2009), and White et al. (2011b).

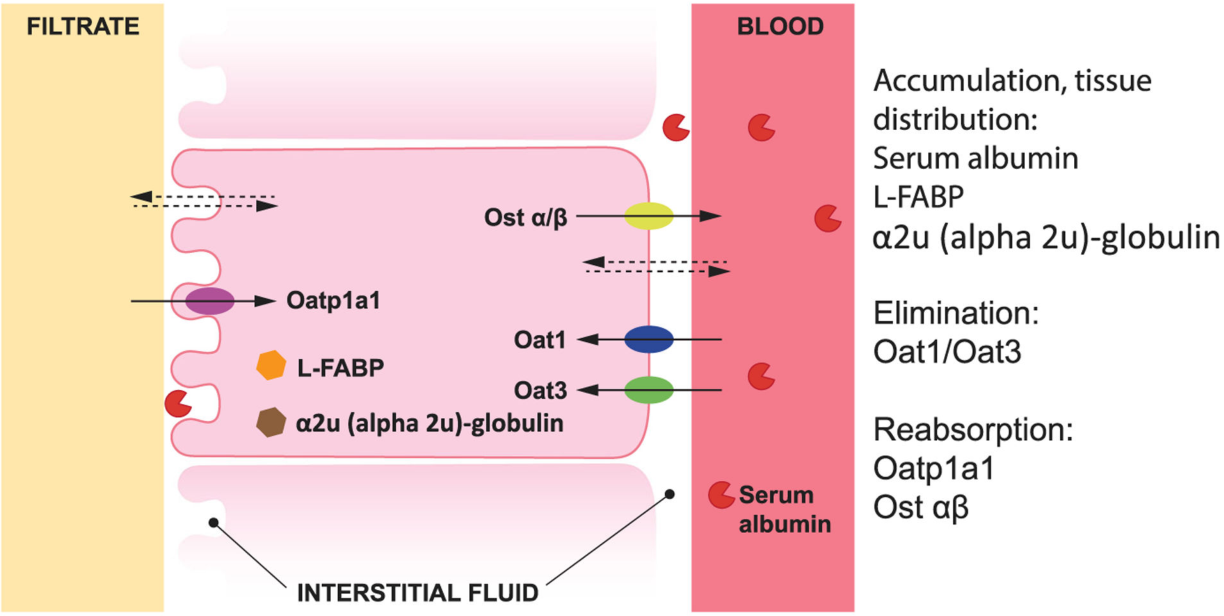

FIGURE 2:

Example of proteins that are known to influence per- and polyfluoroalkyl substance toxicokinetics through binding (which affects tissue distribution and accumulation) and facilitation of membrane transport (which affects clearance and reabsorption). Illustrated for kidney and blood. L-FABP = liver fatty acid binding protein; Oat1 = organic anion transporting 1; Oatp1a1 = organic anion transporting polypeptide 1a1; Ost = organic solute transporter.

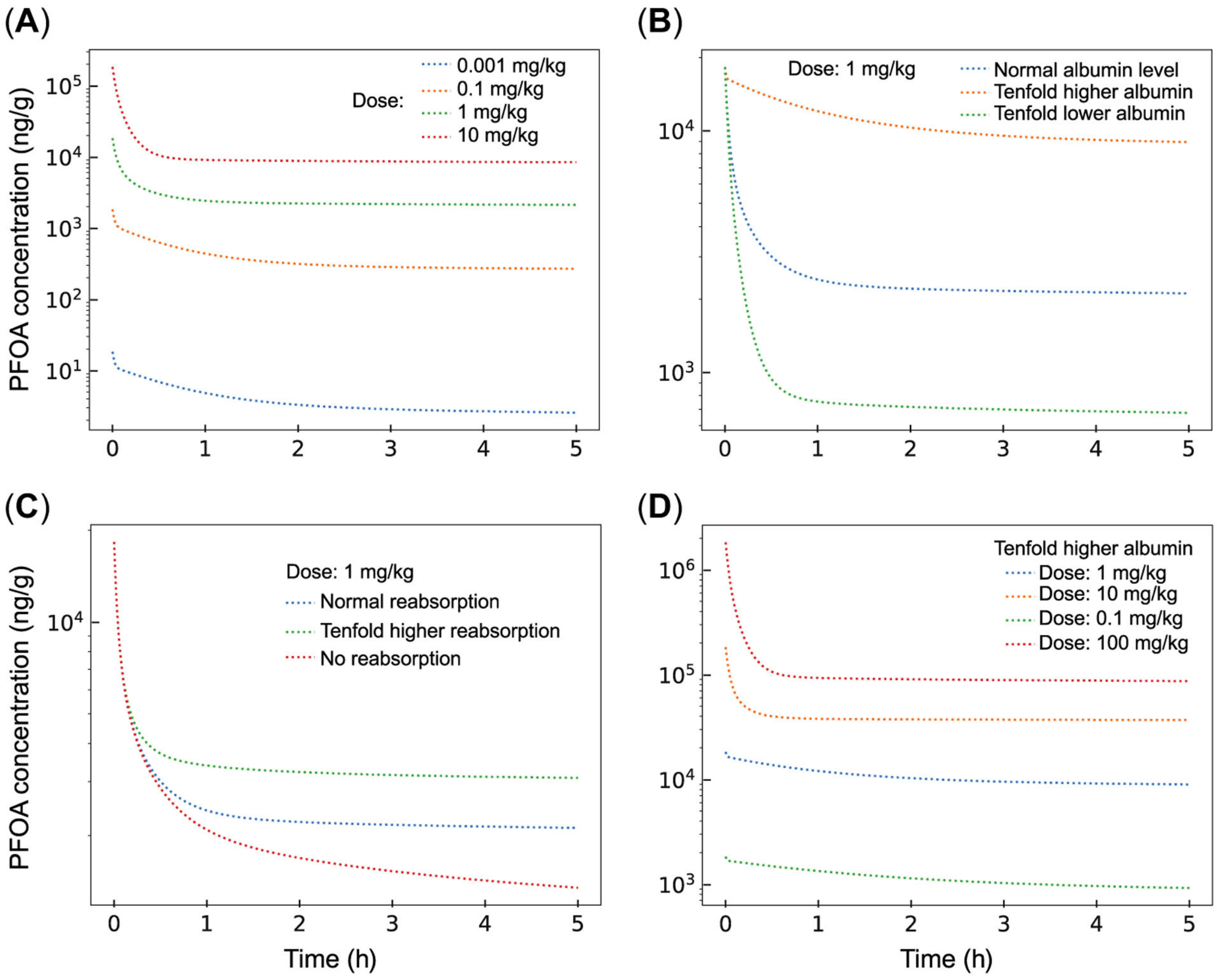

FIGURE 3:

Simulations based on Cheng and Ng (2017), perfluorooctanoic acid (PFOA) toxicokinetic model for Sprague-Dawley rats. (**A**) Effect of dose on initial half-life. (**B**) Effect of higher and lower levels of serum albumin, which binds to PFOA, on serum clearance dynamics. (**C**) Effect of extent of reabsorption in kidney on serum half-life, based on organic anion transporting polypeptide 1a1 activity. (**D**) Effect of dose on elimination kinetics when half-life is longer because of higher albumin binding. Oat1 = organic anion transporting 1; Oat3 = organic anion transporting 3; Ost = organic solute transporter.

TABLE 1:

Per- and polyfluoroalkyl substances serum half-life estimates in rat, mouse, monkey, and humans

	PFBS (C4)		PFHxS (C6)		PFOS (C8)		PFBA (C4)		PFHxA (C6)		PFHpA (C7)		PFOA (C8)		PFNA (C9)		PFDA (C10)		F-53B		GenX	
	F	**M**	**F**	**M**	**F**	**M**	**F**	**M**	**F**	**M**	**F**	**M**	**F**	**M**	**F**	**M**	**F**	**M**	**F**	**M**	**F**	**M**
Rat	0.6–4.0 h	2.1–4.5 h	1.8 d	6.8 d	62–71d	38–41d	1.0–1.8 h	6–9 h	0.4–0.6 h	1.0–1.7 h	1.2 h	2.4 h	2–4 h	4–6 d	1.4–6.4 d	31–55 d	59–75 d	40–80 d			8 h	3 h
Mouse	4.5 h	5.8 h	25–2 d	28–3 d	31–3 d	36–4 d	3 h	12 h	~1.2 h	~1.6 h			16 d	22 d	26–6 d	34–6 d					18 h	20 h
Cynomolgus Monkey	3.5 d	4.0 d	87 d	141 d	110 d	132 d	1.7 d		2.4 h	5.3 h			30 d	21 d								
Human	28 d		5.3–8.5 yr		3.4–5.0 yr		3 d		32 d		1.2–2.5 yr		2.1–3.8 yr		2.5–4.3 yr				15.3 yr			

GenX = hexafluoropropylene oxide dimer acid; PFBA = perfluorobutanoic acid; PFBS = perfluorobutanesulfonic acid; PFDA = perfluorodecanoic acid; PFHpA = perfluoroheptanoic acid; PFHxA = perfluorohexanoic acid; PFHxS = perfluorohexane sulfonate; PFNA = perfluorononanoic acid; PFOA = perfluorooctanoic acid; PFOS = perfluorooctane sulfonate.

TABLE 2:

Fit-for purpose assays proposed in the REACT program

Endpoint of interest	Assay proposed
High-throughput transcriptomics	Metabolically competent human liver cells/MCF-7 (Tempo-Seq®)
Hepatotoxicity	2D HepaRG® cells
Developmental toxicity	Zebrafish embryo assay
Developmental neurotoxicity	Multielectrode array in neonatal cortical cells and neurite outgrowth
Immunotoxicity	Cytokine alterations in human vascular endothelial cells (BioSeek®)
Hepatic clearance	Metabolic clearance in 50 donor-pooled hepatocyte suspensions
Plasma protein binding	Serum protein binding assay using human serum
Enterohepatic recirculation	Qualyst B-CLEAR® hepatocyte transporter assay
In vitro disposition	In vitro disposition in cell lines under study

REACT = Responsive Evaluation and Assessment of Chemical Toxicity.

water

MDPI

Review

An Overview of Per- and Polyfluoroalkyl Substances (PFAS) in the Environment: Source, Fate, Risk and Regulations

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Abstract: The current article reviews the state of art of the perfluoroalkyl and polyfluoroalkyl substances (PFASs) compounds and provides an overview of PFASs occurrence in the environment, wildlife, and humans. This study reviews the issues concerning PFASs exposure and potential risks generated with a focus on PFAS occurrence and transformation in various media, discusses their physicochemical characterization and treatment technologies, before discussing the potential human exposure routes. The various toxicological impacts to human health are also discussed. The article pays particular attention to the complexity and challenging issue of regulating PFAS compounds due to the arising uncertainty and lack of epidemiological evidence encountered. The variation in PFAS regulatory values across the globe can be easily addressed due to the influence of multiple scientific, technical, and social factors. The varied toxicology and the insufficient definition of PFAS exposure rate are among the main factors contributing to this discrepancy. The lack of proven standard approaches for examining PFAS in surface water, groundwater, wastewater, or solids adds more technical complexity. Although it is agreed that PFASs pose potential health risks in various media, the link between the extent of PFAS exposure and the significance of PFAS risk remain among the evolving research areas. There is a growing need to address the correlation between the frequency and the likelihood of human exposure to PFAS and the possible health risks encountered. Although USEPA (United States Environmental Protection Agency) recommends the 70 ng/L lifetime health advisory in drinking water for both perfluorooctane sulfonate (PFO) perfluorooctanoic acid (PFOA), which is similar to the Australian regulations, the German Ministry of Health proposed a health-based guidance of maximum of 300 ng/L for the combination of PFOA and PFOS. Moreover, there are significant discrepancies among the US states where the water guideline levels for the different states ranged from 13 to 1000 ng L^{-1} for PFOA and/or PFOS. The current review highlighted the significance of the future research required to fill in the knowledge gap in PFAS toxicology and to better understand this through real field data and long-term monitoring programs.

Keywords: poly-fluoroalkyl substances (PFASs); toxicology; PFAS health risk; regulatory values

1. Introduction

Widespread surface and groundwater contamination with perfluoroalkyl and polyfluoroalkyl substances (PFASs) has become of great concern in the last few years. PFAS was first realized in the globe through the identification of perfluorooctane sulfonic acid, $C_8F_{17}SO_3H$ (PFOS), in wildlife [1,2].

PFASs have recently received increasing global attention because of their persistence and toxicity in the environment, bioaccumulation potential, and possible adverse health impacts [3]. PFAS are commonly have an aliphatic carbon composition in which hydrogen molecules have been replaced by fluorine completely (prefix: per-) or partially (prefix: poly-) [4]. These compounds are characterized by their highly polar and strong carbon fluorine bonds [5]. They are considered as highly fluorinated surfactants that have been applied in numerous industrial applications and manufactured goods including food packaging, firefighting foams, clothes and protective coatings for fabrics and carpets, electronics and fluoropolymer manufacturing [1,2,5–9]. The most extensively produced and frequently detected PFASs in the environments are perfluorooctanoic acid, $C_7F_{15}COOH$ (PFOA) and perfluorooctane sulfonic acid, C8F17SO3H (PFOS) [1]. PFASs have been discovered in different environmental compartments, including water, sediment organisms, and air [6,10–12].

PFAS has been a serious concern to industry, governments scientists, and even to the public worldwide [13]. It has been detected in various aquatic matrixes, including rain, snow, groundwater, tap water, lakes, and rivers with the C8-based substances PFOS and PFOA typically being the dominating compounds [4,9,14]. PFAS degradation products can be freely mobile in water, soil, and air, and can be extremely resistant to breakdown by different processes. The complexity of measuring PFAS in various media, and the associated unknown risks are among the challenges facing the current regulatory bodies [4]. Typical concentrations of PFASs in water are very low, however, higher concentrations of (mg/L) have been observed in surface and groundwater after firefighting activities closed to fluorochemical manufacturing facilities. PFASs spread worldwide has triggered the governmental concern towards regulating the exposure and spread of PFASs [15,16]. Although there is enough evidence about the negative impacts of PFAS on human and animal health, the scale of the risk imposed by PFAS compounds is not fully understood. The current regulations tend to address the potential risk limit for various wildlife where the PFASs persistence, bioaccumulation potential, and toxicity (PBT) raise a great concern [6]. Several studies have reviewed various aspects related to PFASs fate and behavior in different environments. They also reviewed the sources and occurrence of PFOA in drinking water, toxicokinetic, and health impacts [17–21]. Other reviews on PFASs have discussed different aspects such as environmental biodegradation of PFASs, PFASs removal from drinking water treatment plants, wastewater treatment plants and PFASs transformation in landfills [22–24]. The authors are aware of the developing research concerning PFAS and the many reviews investigating the PFAS human exposure, fate, transport, accumulation, health hazard and guidelines [2,5,16,25–31]. The current mini review investigates the PFAS occurrence in collective all geo-environmental compartments and is the first to collate the various international PFAS standards in one article. The current study reviews existing publications in the field of PFAS and aims to: (i) summarize the recent publication in the field of PFAS and ensure easy access of the research on the occurrence and behavior of PFASs in various environments, (ii) to identify knowledge gaps in the PFAS field, particularly the discrepancies in the current prevailing legislation and practices across various countries, and (iii) to present the key future research directions to better address the PFAS issue.

2. The Developing Trend in PFAS Research

PFAS was first detected in the early 1950s in the form of PFAO and PFOS as a part of the Teflon production process [4]. A few decades later, in the early 1990s, and due to the development of the analytical techniques and instrumentation advancement, PFAS was detected in environment at low concentrations. The investigation on PFASs was evolved in early 2000s when a voluntary phase- out in production of the parent chemical to PFOS was undertaken [32]. Due to the significant development of PFAS production in 2009, more attention was paid to limiting PFAS production whereby many researchers and institutions investigated the source, fate, and impact of PFAS compounds (Figure 1). The related articles published recently were extracted from Scopus based on the following keywords: perfluoroalkyl and polyfluoroalkyl substances, assessment of perfluoroalkyl substances, accumilation toxicity of PFASs, treatment of perfluoroalkyl and polyfluoroalkyl. In total, 122 articles were selected based on their relevancy, scope, and depth of discussion.

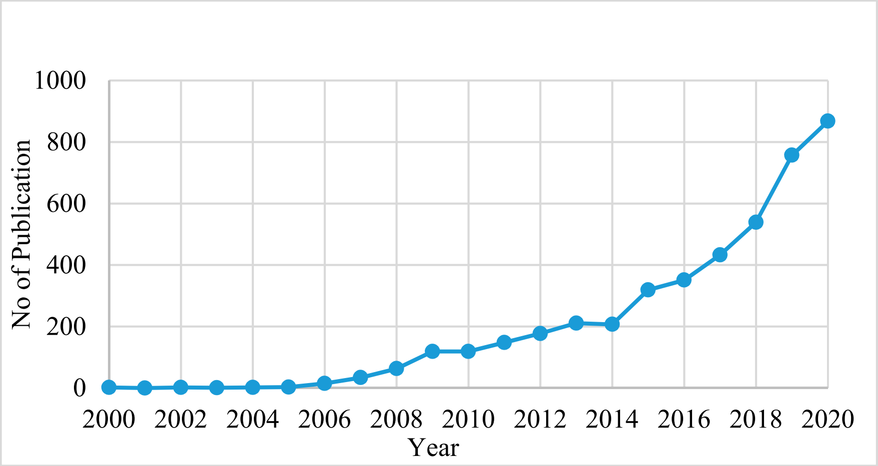

Figure 1. Total number of documents published, concerning PFAS from 2000 to 2020 (data extracted from Scopus; October 2020).

As a result, the research trend reflected by the number of publications concerning perfluoroalkyl and polyfluoroalkyl compounds have been augmented considerably in the last two decades and resulted in a much better understanding of the adverse health effects related to the exposure of PFOA and PFOS [33].

Moreover, it was observed that developed countries have invested much more than other countries on PFAS research, which was reflected by the research funding number of publications as shown in Figure 2. Nevertheless, not all countries share the same concerns and interests due to many various reasons. Some of these reasons are driven by economic factors, where the PFAS is still not on the priority list, as shown in Figure 2. Other factors are associated with industrial and sociopolitical factors, where PFAS form a significant part in industry.

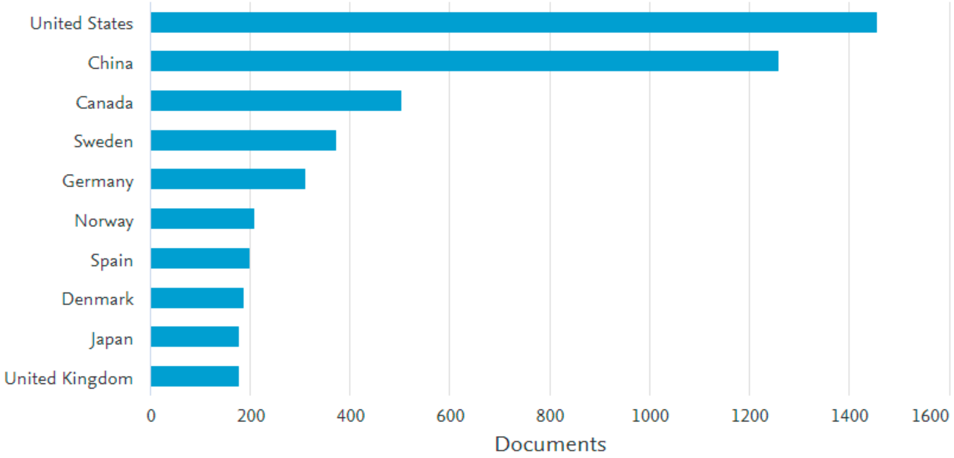

Figure 2. Total number of documents published by top 10 countries concerning PFAS from 2000 to 2020 (data extracted from Scopus; October 2020).

3. PFASs Occurrence and Transformation

Due to the strong C-F bonds in PFASs, they are highly stable and PFASs are unlikely to degrade easily in the environmental matrices [34]. PFASs in the environment has been resulted from several sources. The sources of PFASs in groundwater, drinking water, and surface water could be categorized into (i) point as well as (ii) diffuse sources. Wastewater treatment plants are considered as the most common point sources of PFASs to surface water. Other forms of point sources have been found to have a high impact on surface water in the USA including industrial pollution from PFAS production sites. In addition, it was observed that high concentrations of PFASs can be existed in surface water closed to commercials and military places due to the usage of aqueous film forming foams (AFFF) that contains PFASs. Landfills are considered as important point sources for PFASs in groundwater that comprise PFAS polluted waste in China, and that they could cause a hazard for tap water pollution [35]. Also, in Europe, landfills have not been supported to a significant degree in terms of their capacity for groundwater PFAS contamination. The following subsections illustrate the different occurrence of PFASs in the environments.

3.1. PFAS in Environment

3.1.1. PFASs in Water

The level of PFAS as well as their fate in water bodies have been investigated be many researchers as water represents one of the main pathways for human exposure to [36–41]. The consistent detection of PFAS compounds such as perfluoroalkane sulfonates (PFSAs) and perfluoroalkyl carboxylates (PFCAs) in tap water samples at various locations has raised the concern over their potential health risk [37]. Such detection has been reported in drinking water samples in Europe, China, Malaysia, Thailand, USA, Singapore, Vietnam, and Brazil [4,14,20,36,37,40–48]. Other compounds, including perfluoro hexanoic acid (PFHxA) [38], perfluorooctanesulfonamide (PFOSA) [32], and perfluorinated phosphonic acids (PFPAs) [49], are also among the commonly detected compounds in water samples. This relatively persistent level of PFAS exposure increases with the increased drinking water contamination events where it was estimated that the average human daily PFAS intake ranges from 0.17 to 0.21 ng/kg bodyweight/day for PFOS and PFOA, respectively [50]. The results by Gellrich et al. [50] revealed that short chain PFAS (<8 carbon atoms) were dominant in samples collected from tap water with a maximum level of 42.7 ng/L followed by mineral water and spring water [50]. One of the kay aspects on PFAS level in drinking water is the difference in PFAS level in both treated and source water. An interesting finding by Lu et al. [37] indicated that PFAS concentration was higher in treated water compared with the source water which could be accounted by the potential contamination that may occur whilst treated water moving through the water network and the treatment plant facilities [37]. Moreover, literature showed that various and inconsistent pattern of PFAS compounds was found on many occasions. While PFOA was dominant PFAS compound in tap water samples tested from Shanghai, Beijing, and Nanjing, PFOS was the key PFAS compound in water samples collected from Shenzhen and Hong Kong, accounting for more than 50% of the total PFAS compounds.

Moreover, researchers found mysterious temporal and spatial patterns of PFAO and PFOS across the various events where a comparison of PFAS levels in tap water in various Chinese cities revealed that the PFAS level varied significantly from one city to another with the highest was reported in Shanghai [37]. Moreover, the inconsistent and varied PFAS level was also significant across various countries. Quinete et al. [42] found that, unlike the pattern and the level of PFAO and PFAS level in the USA and Japan tap water samples, PFOS level was higher than PFOA in tap water samples collected in China [42]. An average level of total PFCs of 130 ng/L was measured in tap water samples from Shanghai (China), and a much lower maximum PFCs level was identified in water samples from Toyama (Japan) (0.62 ng/L) [44]. An average of 2000 ng/L was identified in in treated drinking water distribution system at the city of Oakdale, USA. This seems to be a very extreme level of PFAS where a health-based drinking water level of 0.04 g/L was assessed as a protective lifetime exposure through risk assessment exposure [12].

One of the main concerns about PFAS contamination is their persistence and bioaccumulation properties as well as the potential to travel with either water streams or sediments. Traditional treatment facilities seem unable to eliminate PFASs during normal treatment processes [51]. PFAS discharge into water bodies was also reported by Boiteux et al. [47] where river water proved to be impacted by the nearby fluorochemical manufacturing industry. This confirms that discharge of PFOA and PFOS are still detected in nearby industrial and manufacturing facilities. Results showed that river water and sediment samples as well as treated water samples at various stages from the main treatment plant have all showed various level of PFCA compounds coming from the manufacturing industry. Interestingly, PFCA was also detected at sediment samples at 62 km away from the source in almost 50% of the samples [47]

The occurrence of PFASs in surface water is frequently happening across many countries around the world [52]. Researchers have conducted several field studies for different types of surface water to investigate the occurrence and presence of PFASs [2,23,29,53,54]. A previous study was conducted to assess the level of PFASs from different locations in Gangs River, India. Results showed that around 15 types of PFASs were found in the water samples where the highest level detected was for PFHxA and PFBS. In addition, significant relationships were detected ($p < 0.05$) between the different PFASs substances such as PFCAs, PFSAs, PFBA, and PFHxS, indicating chemical binding and co-transport with dissolved oxygen carbon (DOC) in fresh and seawater. Consequently, assessed the pollutants concentration and spatial distribution of PFASs in Shuangtaizi Estuary, China. Results showed that the Shuangtaizi Estuary was in general polluted by PFASs. The total concentration of PFASs varied from 66.2 to 185 ng/L and from 44.8 to 209 ng/L in surface and bottom water of the Shuangtaizi Estuary, respectively, where the maximum concentration was reported for PFBS and PFBA. The level of PFASs in different environmental matrices was tested including surface runoff water rain, snow, and lake water in an urban area, to identify the sources of PFASs to urban water bodies [36,39,40]. Another research conducted by Yin et al. [55] discovered a significant temporal variation of PFASs compounds level over 12 months period due to the seasonal and climatic dry and wet conditions. Moroer, PFASs concentration was a function of the chain length where the level of short-chain compounds including PFBS, PFHxA and PFHpA tend to be highly influenced and decreased by the wet conditions. On contrary the level of long-chain PFASs compounds was more stable in both wet and dry conditions. These findings provide a good understanding to the leachate of PFAS compounds from point source pollution as landfills and treatment plant. The leaching of long chain is more likely controlled by the partitioning effect whilst the short chain leaching is influenced by the climatic conditions [55].

Another main finding in the field of PFAS in water is the variation of the PFAS where PFOA was the major compound with an average concentration of 35% of the total PFASs levels, in all environmental matrices investigated. In addition, the concentrations, and relative substances of PFASs in surface water were comparable to the concentrations found for urban lakes. Surface water leads to PFOA pollution in urban lakes. A sampling campaign was conducted in different seas in China in 2012. The results revealed that the higher concentration of PFAS was detected in the South Yellow Sea, where FTOH was the predominant substance, contributing 92–95% of the total PFAS [2].

3.1.2. PFASs in Soil

PFAS was detected in soil at various concentrations due to the reach out from various pollutions sources where PFAS compounds retain in soil due to sorption, partition and other complex reaction [56]. Table 1 shows the range of PFAS concentration in soil. The application and the reuse of sludge from wastewater treatment plants in farmlands is one of the main sources that contribute to soil contamination [57]. Other sources could be due to the degradation of fluorotelomer-based materials that lead to the release of PFCAs [43], precipitation, and water irrigation [58]. PFAS compound in soil in coastal areas can be emitted from direct sources which could level the PFAS concentration up to around 8–50 $\mu g\ kg^{-1}$ soil as was reported in Chinese soil [45] which is somehow higher than the proposed PFCs in soil proposed by the USEPA (6 mg/kg for PFOS and 16 mg/kg for PFOA). One main concern about PFAS in soil is the potential PFAS release and carryover by plants as well as the possible PFAS leaching to the underneath soil layers and the groundwater. This carryover of PFOA and PFOS to the plant was evident where the PFAS level in plants was proportionally related to the PFOA/PFOS in the soil [59]. PFCs uptake from contaminated soil by crops was reported [60], where samples from rye grass, grain, and potatoes showed high potential of PFCs transfer from soil to crops [60]. This resulted in proposing a preventative PFCs limit of 100 ng/g dry soil in sludge to be reuse for farming purposes as to limit the potential transfer of PFCs from soil to plants and crops [59]. Moreover, the potential leaching of PFAS from soil through vadose zone is another threat that requires more attention and understanding where insufficient data about in-situ soil remediation and contaminants leaching to the groundwater are available [60]. The development of PFAS compounds in the soil system is complex since PFAS compounds can attain both hydrophilic and hydrophobic characteristics [56]. While the transfer of PFAS from soil to plant roots undergo through diffusion and sorption onto roots, there are still insufficient details about the PFC transfer rates in various crops and vegetables [59]. This result was confirmed [59] where they found the straw and grains of maize plants have the same carboxylic and sulfonic functional groups as in the contaminated soil referring to a direct correlation between soil and crops PFASs contamination.

PFASs toxicity their impact on soil microorganisms is among the other factors that can deteriorate the soil quality. Research found that the PFCs can negatively affect the soil functionality where it may disturb soil enzyme activity as well as change the microbial availability and damage the cellular structure [61–63]. The same result was confirmed by Sun et al. [64] as the soil contaminated with PFASs compounds had less bacterial diversity [64]. PFOA and PFOS are the dominating compounds reported in soil where their concentration ranged from <1 to around 13,000 ng/g in soil [56]. The fate of PFASs in the soil is a function of many parameters including soil pH, soil structure, clay content, organic matter content (OM), PFAS characteristics (long versus short chain), and climatic conditions [61]. OM seems to be the most significant controlling factor determining the PFAS toxicity level where the PFAS toxicity is inversely proportional with the soil OM content [34,65]. Additional research on PFAS adsorption and migration from soil to the groundwater and how this can migrate with the groundwater is still a research gap needs more investigation and modelling to account for the various PFAS concentrations in various groundwater conditions [60]. Contaminated soil with PFAS is a challenge since there is no definite remediation strategy to address the in situ PFAS remediation. Although soil stabilization using various reagents such as clay and Portland cement seems to be a promising technique for soil remediation, it does not provide an elimination for PFAS where it does not remove PFAS permanently [60]. Finally, the PFAS uptake by plant poses a direct human risk where the food chain represents a main risk pathway. Therefore, a toxicological risk assessment addressing the maximum allowed levels of 1.5 and 0.15 µg/kg body weight as TDI µg/kg for PFOS and PFOA, respectively, were identified by the European Food Safety Agency (EFSA) as a function of the respective tolerable daily intakes (TDI) of the compounds [59,66].

PFASs compounds are soluble in water and have the potential to leach down to the groundwater particularly in areas with potential source pollution like landfills and treatment plants. PFASs occurrence and leaching was reported by many researchers around the world [22,55,67–69]. The potential PFAS leaching could be alarming in many cases where PFASs were detected at large depths (15 m) below ground [70]. Yet, the leaching speed and behavior vary from one PFASs to another which depends on the soil binding, retardation and adsorption capacity [56]. The leaching characteristics of PFASs compound is a function of the chain length where short chain is more mobile than long ones. An analysis of the landfill leachate from 27 landfills in Australia was investigated by Gallen et al. [22]. Interesting findings presented in their study showed that the landfill leachate was significantly different from one landfill to another with an average PFASs of 1700 ng/L and a maximum PFAS level of 25,000 ng/L [22]. In contrast, the reported PFAO range in USA landfills was ranging between (7280–290,000) ng/L compared with 214,000 ng/L in China [36]. Nonetheless, these PFAS concentration are highly likely to vary due to the heterogeneous nature of waste dumped in landfills as well as the varied PFAS content in the generated landfilled materials. Operating landfills receiving municipal waste had much more PFAS level than closed ones and the leachate from landfills with construction and demolished materials seems to leach more PFAS than municipal landfills. Another study investigated the leachate from 11 landfills in USA and found that PFAO was detected in all samples [70]. Table 2 presents the level of various PFCS and PFAS compounds in leachate and compare the PFASs in water and solid. It can be seen that PFASs levels vary from one compound to another as a function of chain length and climatic conditions as illustrated in the previous sections. The risk associated from the landfill leachate is the potential volumes leachate generated particularly in wet climates, which contributes to the groundwater contamination. The total leachate volume in the USA was estimated to be around 61 million m^3 with around 80% coming from landfills [70]. Meanwhile, the leachate mass of $\sum$PFA in China was estimated by around 3 ton per year with the landfill leachate contribute to around 35% of this quantity [35]. Interestingly, analysis of leachate from young landfills showed much higher PFAS concentration in many occasions confirming the fact that the complexity and persistence of PFAS compound has been developed in the recent years where more frequent PFAS containing materials are in use [67,70]. The uniqueness of the landfills associated with its design capacity, climate, age, engineering, dumbed materials and frequency and other factors made it hard to predict the amount of PFAS leachate in various landfills where ad-hoc studies to be conducted. The results from various areas across the globe showed significant variation of PFAS leachate from one country to another where a maximum was reported in Australia (25,000 ng/L). This was evident while the leachate was significantly lower in Norway (590 to 757 ng/L), Germany (<0.37 to 2509 ng/L), and China (146 to 4430 ng/L) [35]. In conclusion, although the phasing out of PFAS materials and the ongoing effort to eliminate the PFAS release in the environment, yet there seems to be a need to consider more adaptation strategies dealing with PFAS risk. The increasing evidence of PFAS in newly designed and operated landfills indicates the potential exposure to higher leaching risk with greater PFAS concentrations is leaching to the environment is growing [70].

Table 1. Ranges of PFAS concentration in soil.

Header	**Å. Høisæter, et al. [71] ng/g PFASS in Soil at Various Depths**				**Cai et al. [63] ng/g Dry Weight in**	**Chen et al. [69] ng/g Dry Weight**	**Cai et al. [61] ng/g Dry Weight**	**Gao et al. [72] ng/g Dry Weigh from 32 Samples**	**Wang et al. [73] The Mean Values ng/g Dry Weight**	**Liu et al. [74] ng/g Dry Weight**	**Chen et al. [75] ng/g Dry Weight**	**Dalahmeh et al. [76] ng/g Dry Weight**	**Armstrong et al. [77]**
	0–1.0 m	1–2 m	2–3 m	3–4 m	Dry soil in China	Tot.PFC 0.34–65.8			PFAS in soil at varying distances				
PFOS	500–3000	1000–6500	1000–3500	1000–1200	130	70.5	8.6–10.4	0.06	2583,	87	0–2	0.6–3	23
PFOA	NA	NA	NA	NA	NA	93	3.3–47.5	0.32	50	0.3–8	63	0.5–0.9	24
PFHxS	NA	NA	NA	-	NA	61	NA	0.19	36		65		
∑PFCs	NA	NA	NA	NA	NA	-	NA	-	NA	99	NA	8	126–809
PFHxA	NA	NA	NA	NA	NA	NA	NA	0.09	NA	NA	NA	0.2–0.5	8
PFAA	NA	NA	NA	NA	NA	NA	NA	1.30–913	NA	NA	NA	NA	NA
PFBS	NA	NA	NA	NA	NA	NA	NA	0.05	NA	NA	NA	NA	NA

Table 2. Ranges and mean concentration of individual PFCs in landfill leachate.

Header	**Gallen et al. [22] (ng L^{-1})**		**Busch et al. [68] (ng L^{-1})**	**Herzke et al. [78] (ng L^{-1})**	**Clarke et al. [79] (ng L^{-1})**	**Yin et al. [55] (ng L^{-1})**	**Benskin et al. [80] (ng L^{-1})**	**Eggen et al. [81] (ng L^{-1})**	**Robey et al. [69] (ng L^{-1})**	**Fuertes et al. [82] (ng L^{-1})**		**Eggen et al. [81] (ng L^{-1})**		**Huset et al. [83] (ng L^{-1})**	**Garg et al. [84] (ng L^{-1})**
	landfill (>50% SW)	landfill (>50% C&D)	Compounds in landfill leachates	Coated textiles, Teflon waste, fire-fighting foam, papers, and furniture		Leachate from CW outlet system (Max. level)	Municipal landfill leachate	Municipal land fill leachates	Foam produced via the bubble aeration of landfill leachate	Raw Leachate in MSW landfill	Treated Leachate in MSW landfill	PFCs analysis-untreated leachate Water	PFCs analysis -untreated leachate Particles	Leachates from six landfill	Manufacture and disposal of electric and electronic products
PFOS	300	1100	235	570	187	439	4400	2920	104	25	NA	2920	34	56–160	128,670
PFOA	510	1200	926	9500	516	3457	1500	767	951	590	520	767	4	380–1100	118.3
PFHxS	940	3700	178	-	143	308	190	281	2058	630	870	281	ND*	120–700	133,330
PFDS	-		-	-	NA	0.72	63	<14	ND			-	-	0–16	-
PFHxA	1300	5000	2509	-	697	868	2500	757	2178	65	77	757	ND	270–2200	76
PFHpA	360	760	280		NA	486	690	277	454	-	-	277	ND	100–2800	9
PFNA	29	98	80	-	62	100	450	539	64	-	-	539	ND	19–140	8
PFDA	22	46	51	-	NA*	27	1100	75	87	-	-	70	ND	0.3–64	8
PFAA	NA	NA	NA	-	-	55	-	-	-	-	-	-	-	-	-
PFBS			1350	NA	112	1916	190	-	-	-	-	<5	ND	280–2300	-
PFPeA										11	325	-	-	-	

PFDA perfluorodecanoic acid, PFBS nonafluorobutane-1-sulfonic acid, PFHpA perfluoroheptanoic acid, PFHxS perfluorohexane sulfonate, PFNA perfluorononanoic acid. ND*: not detected, NA*: not analysed.

4. PFAS Treatment and Clean Up: Challenges and Achievement

Due to the persistence nature of the PFAS compounds, landfills and sewage treatment plants are highly likely to be a potential point source of PFAS emissions. Although the rapid advancement in PFAS testing and detection, yet the available standard analytical methods still short and there is very little experimental data detailing the physicochemical properties and partitioning constants of PFAS [19]. This place the treatment process of PFAS under the stress of producing precise and consistent outcomes. The literature presents various treatment methods of PFAS in various environment. Immobilization and plasma arc destruction are among the recommended methods to irreversibly transform PFAS waste. Some studies support the utilization of high temperature incineration as well as the usage of plasma destruction of PFAS in waste. It may be possible for certain types of waste. However, in the absence of regulation, there is no specific method to guarantee the universal adoption of such safer methods of disposing of PFOS wastes. In European wastewaters, it was found that PFOA is the most commonly found compound [84]. Conventional processes of wastewater treatment were found to be ineffective in removing of PFOA [85]. Other studies have recorded higher concentrations of PFOA in wastewater effluents than influences, possibly due to the transformation of the compounds of its precursor [21]. It was revealed that the degradation of precursor compound substances is a major supplier to environmental PFAS pollution. Thus, this part focuses on the treatability of PFAS compounds via conventional and modern water treatment processes. A comprehensive revision is judgmentally required to have a clear understanding on the transformation, migration and treatment of these substances in ecosystem and their potential influence on the secondary formation of PFOS and PFOA [86]. In order to eliminate and/or degrade PFAS, different pre-treatment methods have been tested in terms of their efficacy [19,20]. Some of these elements can theoretically be implemented as either a post-treatment or pre-treatment method with controlled aquifer recharge as summarized in Table 3.

Table 3. Potential Treatment technologies of PFAS.

Mechanism	Treatment Process
Destructive Treatment	Advance oxidation processes Electrochemical oxidation Incinerations Sono-chemical Biodegradation Photolysis
Non-Destructive treatment	Adsorption Ion exchange Fractionation

A detailed review summarizing the sorption mechanism along the sorption coefficients and capacity of PFAS on sediments is available [11]. Adsorption via activated carbon and ion exchange resins have been widely employed especially for pump-and-treat remediation to extract PFAS from polluted groundwater [52]. Compared to others, the use of GAC for PFAS removal has been recommended as a cheaper process, and it is the most recognized treatment technology for the groundwater contaminated by PFOS and PFOA [19]. Although the removal efficiency of polyfluoroalkyl substances by granular activated carbon was >90%, yet the sorption kinetics are normally faster for longer-chained PFAS [87]. For example, Kucharzyk et al. [19] reported that GC, which is optimized and applied effectively for the removal of PFOS, may not be appropriate for the removal of shorter-chained PFAS.

There is a risk that shorted-chained PFASs are more likely than their longer chain counterparts to split through a GAC medium. Otherwise, given the highly persistent existence of PFAS, stockpiling of spent GAC would turn out to be a serious hazardous waste management concern. Storage space is often restricted in MAR systems, and when leaching to groundwater, the disposal of PFAS polluted

GAC at landfills can present a secondary contamination source to the ecosystem near the landfill site. Methods that can effectively kill PFAS are therefore highly desirable in both polluted water and stockpiled GAC. Other researchers have shown that nanofiltration can successfully extract PFOA from a spiked sample of groundwater [47]. In their study, three different levels of PFOA, including 5, 50, and 100 µg/L, were examined, and it was noticed that the remediation effectiveness was higher at a high PFOA level. At higher PFAS level, RO process could contribute to about 99% removal of PFOS with an initial concentration of 500 to 150,000 µg/L and a combination of nanofiltration TOGOTHER WITH reverse osmosis (RO) can achieve 99% PFOS removal and 90–99% PFOA (10,000 µg/L) removal throughout four days of treatment [88]. Nevertheless, the technique was not capable to ensure that the treated effluent was less than the recommended guideline values, even with high removal acacias (99%). Using RO and nanofiltration membranes showed that accumulation (fouling) of PFAS cause a substantial reduction in flux in the filtration process [88]. Unfortunately, the main drawbacks of nano-filtration processes are the low water recovery (75% to 80%) and the existence of high concentrations of inorganic substances comprising magnesium calcium, and silica in groundwater [89]. This provides an indication on the volume of brine water produced which also needs additional remediation before its final discharge. Table 4 illustrates the performance of numerous selected treatment technologies for PFAS at the laboratory-scale.

Table 4. Performance of numerous treatment technologies for PFAS (laboratory-scale).

Process	Treatment Mechanism	Operation Conditions	Performance	References
UV-Fenton	Oxidation	30.0 mM of H_2O_2, 2.0 mM of Fe^{2+}, pH 3.0. and 9 W UV lamp (max = 254 nm)	>95% PFOA destruction from 8.2 mg/L and defluorination efficiency of 53.2%	[88]
Oxidation	30.0 mM of H_2O_2, 2.0 mM of Fe^{2+}, pH 3.0. and 9 W UV lamp (max = 254 nm)	PFOA treatment >95% from 8.2 mg/L while defluorination effectiveness = 53%	Removal efficiency 100% (PFOA 559 mg/L)	[90]
Oxidation	Light-activated persulfate at 50 mM & radiation of 4 h of	Removal efficiency 100% (PFOA 559 mg/L)	73% removal efficiency of PFOS throughout 120 min	[91]
Sonolysis	PFOS level from 65 µg/L to 13,100 µg/L) were treated through ultrasonic at frequency 505 kHz and power density 187.5 W/L).	73% removal of PFOS within 120 min	55–98% removals for different analyzed PFASs. Ozonation can create potentially toxic transformation products	[92]
Oxidation	Tested for 18 analyzed PFASs3 h of ozonation	55–98% removals for different analyzed PFASs. Ozonation can create potentially toxic transformation products which needs to be investigated in future research.	Adsorption capacity 41.3 mg/g of PFOA and 72.2 mg/g of PFOS	[93]
Adsorption	10 mg/L of PFOA; surface area: 534 m^2/g; time of equilibrium 24 h; pH 5	Adsorption capacity 41.3 mg/g of PFOA and 72.2 mg/g of PFOS	Adsorption capacity 510 mg/g of PFOA	[94]
Adsorption	700 mg/L of PFOA; surface area: 1539 m^2/g; time of equilibrium 24 h; pH 7	Adsorption capacity 510 mg/g of PFOA	Adsorption capacity 166 mg/g of PFHxA	[95]
Ion exchange using IRA 67	Particle size: 3–1.2 mm; 120 mg/L of PFHxA; time of equilibrium 12.5 h; pH 4	Adsorption capacity 166 mg/g of PFHxA	Adsorption capacity 2390 mg/g mg/g of PFOS	[96]
Ion exchange using IRA 67	Particle size: 3–1.2 mm; 200 mg/L of PFOS; time of equilibrium 20 h; pH 3	Adsorption capacity 2390 mg/g mg/g of PFOS		

In the degradation of PFOA and PFOS at ppm (mg/L) levels, methods such as advanced oxidation processes (AOPs) using ozone (O_3) and H_2O_2/Fe^{2+} were not effective. At 254 nm, direct UV irradiation was not capable of removing PFOA [84]. At relatively low temperatures (e.g., 40 °C), reaction rates were low and activation at higher temperatures was needed to speed up the reaction [19]. A functional and scalable approach for treating PFAS appears to be sonochemical therapy. For the removal of PFASs from water, AOPs based on heterogeneously catalyzed ozonation were used. Various combinations of ozone, a catalyst and persulfate were performed in laboratory-scale ozonation experiments. These combinations showed high removal efficiency, using all three parameters [97]. In the pilot-scale setup, within three hours of treatment, the concentrations of all 18 analyzed PFASs were reduced significantly. Given that the assessed ozonation treatment is already commercially available for large-scale applications today, it could easily be used in current water treatment trains, but ozonation will create potentially harmful conversion products that will need to be explored in future research. It promises to be used to decrease PFAS levels in PFAS-loaded sorbents as a destructive tool (e.g., spent GAC) [98]. Hydroxyls radicals are normally generated within the bubble from the cleavage of H_2O and O_2 to react with or abolish the pollutants. In another study, sonochemical treatment using a pilot-scale high-power sonicator was carried out for the treatment of 2,4,5-trichlorophenoxyacetic acid [99]. It can be concluded the sonochemical treatment process was effective in removing organic compounds (>90%) within a very short duration (a few minutes). Although the sonochemical technique appears promising for the large-scale treatment of PFAS contaminated products, incomplete PFAS destruction is currently viewed as a disadvantage. A recent study showed that 6:2 fluorotelomer sulfonate was less susceptible than perfluoroalkyl analogs (PFOA and PFOS) to sonochemical destruction and decreased the defluorination rate with a decreased degree of fluorination [100]. Given the recent focus on integrating deferential PFAS treatment processes in treatment trains in order to optimize the overall efficacy of PFAS destruction [98] In order to optimize the overall destruction of PFAS in the polluted media, it seems important to investigate the potential pairing of sonochemical treatment with alternative methods.

Table 5 summarizes the performance of various type of adsorbents in treating per- and polyfluoroalkyl substances. As a conclusion based on the treatment techniques performance investigated previously and partially summarized in Tables 3–5, traditional biological processes of wastewater treatments were found ineffective in removing of PFOA [85]. Due to the exceptional chemical futures of PFAS including high solubility, surfactant property, and thermal stability, various traditional and well-established treatment processes, including chemical oxidation, air stripping, and thermal treatment are invective in treating PFAS [19]. Similarly, Page et al. [21] indicated that numerous techniques like adsorption via granular/ powdered activated carbon, ion exchange resins, reverse osmosis, membrane filtration, advance oxidation techniques, and sono-chemical decomposition have been investigated for the treatment of PFOA and PFOS from water and wastewater. Accordingly, among several physical and chemical treatment processes, adsorption process has been comprehensively tested and have shown to be effective methods for eliminating PFASs from water. It can be observed that the adsorption capacity and treatment performance using adsorption process were increased when the surface area increase [19]. Yet, in this adsorption process, the pollutant will be transferred from liquid phase to solid waste which will need to be managed as a hazardous waste. The main concern and limitation of RO and nanofiltration membranes is the fouling where PFAS cause a substantial reduction in flux in the filtration process [88]. Also, the low water recovery and the existence of high concentrations of inorganic substances comprising magnesium calcium, and silica in groundwater [89]. Yet, the brine water produced also needs additional remediation before its final discharge. Advanced oxidation processes are promising and have high potential in the removal of PFASs from water. AOPs were used recently at laboratory-scale and showed high removal efficiency, using all three parameters [97].

Table 5. Summary performance of various type of adsorbents in treating Per- and Polyfluoroalkyl Substances.

Sorbent	Adsorbate	Operation Conditions	Adsorption Capacity	References
Clay minerals (surface area: 67.52 m^2/g)	PFOS	Initial concentration of adsorbent 400 mg/L; pH7; concentration of adsorbate 0.2 mg/L	0.29–0.31 mg/g	[101]
Kaolinite (surface area: 11.9 m^2/g)	PFOS	Initial concentration of adsorbent 5000 mg/L; pH7; concentration of adsorbate 0.95 mg/L	0.08 mg/g	[102]
Alumina (surface area: 88.6 m^2/g)	PFOA	Initial concentration of adsorbent 10000 mg/L; pH4.3; concentration of adsorbate 0.1 mg/L	0.16×10^{-3} mg/g	[63]
Porous graphite (surface area: 2870 m^2/g)	PFOS	Initial concentration of adsorbent 100 mg/L; pH5; concentration of adsorbate 100 mg/L	1240 mg/g	[103]
Biochar from maize straw (surface area: 7.21 m^2/g)	PFOS	Initial concentration of adsorbent 200–1200 mg/L; pH7; concentration of adsorbate 100 mg/L	91.6 mg/g	[104]
Chitosan (surface area: 2870 m^2/g)	PFOS	Initial concentration of adsorbent 1350 mg/L; pH3; concentration of adsorbate 50 mg/L	645 mg/g	[105]
Zeolite) NaY80 surface area, 780 m^2/g	PFOS	Initial concentration of adsorbent 1000 mg/L; concentration of adsorbate 150 mg/L, Particle size: 3–1.2 mm	114.7 mg/g	[87]
Activated carbon from leaf biomass	PFOA PFOS	Modified activated carbons (AC-H_3PO_4) produced from leaf, uniform particle size of >64 µm	159.61 mg/g 208.64 mg/g	[106]
Modified silica	PFOS	Surface area 650 m^2/g, Particle size: 250–450 µm, Pore volume: 1.03 mL/g	55 mg/g	[107]
Boehmite	PFOS	Surface area 299 m^2/g, Average particle size 37.02 µm	0.1529 $\mu g/m^2$	[108]
Alumina nanoparticles	PFOS	Surface area: 83 m^2/g, Particle size: 13 nm	At 30 °C: 589 mg/g, at 40 °C: 485 mg/g, at 50 °C: 447 mg/g	[109]

5. The Key Knowledge Gaps and Future Research

PFOA and PFOS are the most well-known and well-studied PFASs, with an average removal half-life (t1/2) of 5.4 years and 3.8 years, respectively [7]. In order to ensure that the tested results represent real PFAS levels in the examined media, field sampling and laboratory hygiene procedures are important. With many sampling tools used in field and laboratory operations already containing PFAS, the process of sampling and PFAS testing remain uncertain and need lots of effort to alleviate the uncertainty involved. Figure 3 shows the molecular structure of PFOS and PFOA.

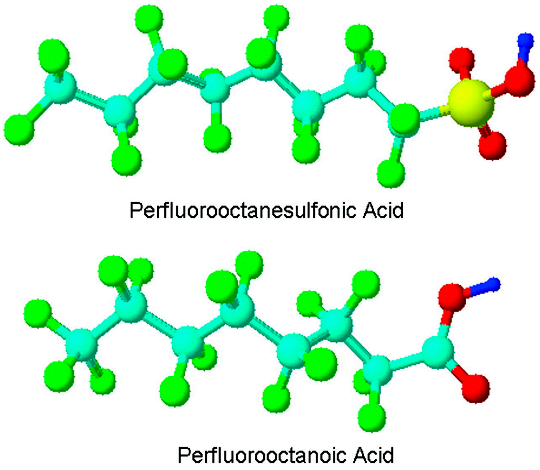

Figure 3. Molecular structures of two representative PFAAs: PFOS and PFOA [8].

Via their fact sheet collection, the Interstate Technology and Regulatory Council (ITRC) outlined site characterization, sampling safeguards, and analytical process concerns and choices. However, for the study of PFAS in surface water, wastewater, non-potable groundwater, and solids, there are currently no validated standard EPA methods [4]. Some US laboratories are applying adapted approaches based on EPA Method 537 for non-drinking water samples. These updated approaches do not have clear sample selection or analytical criteria and have not been checked or analyzed routinely for data quality [110]. As traditional water treatment techniques are unable to effectively remove PFASs, novel treatment methods are urgently needed to remove PFASs in water [92]. Although the intensive effort of phasing out many PFAS products with enforcement of alternative chemical production are in place in many areas across the globe, the risk of PFAS exposures due to the uptake and accumulation in the various media such as ocean and marine food chains as well as groundwater contamination represents a great challenge due to the complexity of the impact timescales [111]. The development and the propagation of the PFAS sites with increased possible exposures to newer PFASs have not been well defined yet.

Despite the agreed health impacts of PFAS particularly on aged or early aged groups, environment, no enforceable national drinking water limits and guidelines are in place in many parts across the globe [33]. There is still limited knowledge around the other PFAS substances. However, there is a growing evidence that these new detected compounds could have same potential on human health and may pose similar risks to human and the environment [33,112]. The use of engineered pre-treatment or post-treatment approaches must be based on a 'fit for purpose' definition and carefully combined with the planned water end use concept in order to make sure that both human and environmental health threats are properly managed and treated [21].

Another main challenge in PFAS potential health impact with considerable complexity was reported by Gebbink et al. [112]. They found that in samples collected from food, not only the PFOS and PFOA levels were overestimated by an order of magnitude, but also there is still a knowledge gap in identifying the precise percentage of these precursors can contribute to human PFCA exposure since the exposure pathway remains undefined [112].

In order to evaluate the development of degradation products and potentially undesirable by-products to track the occurrence of compounds in the gas process and to demonstrate the efficacy of treatment for other types of pollutants and to apply them to different types of water, further research is required. Whilst many studies demonstrate the link between PFAS exposure and the deteriorated immune system particularly in children, there is little evidence to map other health impacts, including cancer, as it is only noticeable in areas with exceptionally high exposures, with inadequate data to correlate these exposures to PFAS with neurodevelopment [111].

5.1. Risks Associated by PFAS

The developing use of the PFAS in various commercial and industrial sectors including aqueous fire-fighting foam, disposable food packaging, furniture, carpets, cookware, water treatment and many others poses a potential risk to the environment [15,16,49]. Like many other contaminants, PFAS can accumulate into the environment by either a direct or indirect pathway [113]. Direct sources and PFAS contamination released from different industries including wastewater treatment plants, sludge disposal, and landfill sites [4]. There are more than 4000 bioavailable PFAS compounds in the globe, however, the toxicity values for most of these compounds are still poorly understood with only few PFAS compounds have defined toxicity values and level [32]. Risk assessment aims at developing health-based guideline levels upon intensive review of PFAS toxicological level that cause harm to humans. PFOS was viewed as Persistent Organic Pollutant (POP) by the Conference of Parties, Stockholm Convention in 2009 where EPA has characterized PFOA as a "likely carcinogen" and its use was restricted [98]. Unfortunately, this PFAS toxicity level is still poorly understood and this has created the need to develop defined hazardous and risk registry for the PFAS toxicity values. This seems to be among the great challenges for the majority of PFAS compounds [53].

Human Exposure Pathways

The first ever published data about widespread PFOS in the environment was reported in 2001 by [6] where PFOS was found in fish and birds tissues as well as marine mammals. The finding revealed that the level of PFOS in animals is proportionally related with the population density and the industrialized activities. As such animals in these areas have much higher PFOS level than those live in remote marine areas [6,30]. Another pathway was found through the food chain where fish eating animals such as mink and bald eagles proved to have greater levels of PFOS than in their diets. Another toxicity realization of PFAS was introduced when PFAS compounds were reported in blood samples from various samples around the globe [32]. Potential health risks associated with PFAS exposure and the concern regarding their bioaccumulation indicated the tendency towards the potential exposure through various media and various sources. A study of consumer exposure using a scenario-based approach to PFOS and PFOA conducted by Trudel et al. [30] revealed an everyday exposure to PFAS is taking place in many countries resulting in the long-term uptake of PFOS and PFOA of 3–220 and 1–130 ng per kg body weight per day, respectively [30]. Many health impacts including cancer, liver damage, and immune system failure have been linked to the PFAS [113,114]. Moreover, whilst the routes of PFAS exposure remains somehow unclear, research agrees that diet is a potentially main source and well-established research suggests that PFOA is absorbed via inhalation and ingestion [30,115].

Human exposure pathways for PFOS, PFOA, and other PFAS related substances through various routes including drinking water, indoor polluted environment, food chain, long term contact with industries that produce PFAS compound including food packaging or cookware, breast milk, airborne dust, and air [1,116]. All of which result in cumulative uptake and PFAS build up [18,23,29]. This persistent exposure to PFAS can be poorly reversible due to slow elimination kinetics as well as the ongoing build up regardless of its magnitude and bioaccumulation potential [23]. China, for example, has witnessed a surge in PFAS release in the last decade due to the evolving industrial activities. The release of PFOS in China from industrial sources in 2010 was estimated by about 70 ton which is six times greater than the reported PFAS release in 2008 [20]. Table 6 shows the PFAS source- pathway and receptors.

Table 6. PFAS Source- Pathway and receptors.

Sources	Exposure Pathways	Receptors
Industrial and wastewater effluents Packaging Consumer products Landfills Fire-fighting foams	Soil Biosolids Dust Sediment Surface water Groundwater Drinking water Biota (including foods)	Ecological Aquatic Benthic Terrestrial Avian Human

Inappropriate treatment and disposal of waste and wastewater proved to pose significant PFAS risk and contamination [110]. Indirect PFAS generation and contamination take place through the transformation of perfluoroalkyl precursors and the breakdown of perfluoroalkyl-based products [116]. Both direct and the indirect PFAS generation can pose high contamination and risk to the surrounding environment including air and drinking water contamination, food poisonous where PFAS have been identified as a potential threat to public health [117]. The great effort made to counter the propagation of PFAS levels in various receptors resulted in PFOS and PFOA declines by 32% and 25%, respectively, in early 2000. Yet, other PFAS compounds continue to increase, suggesting various and more strict measures to quarantine the spread of the PFAS compounds [49]. This was evident through a long monitoring program of PFAS concentration in human blood in Norway (1977–2006) which resulted in the conclusion that PFAS levels increased by more than nine times in men aged between 40 and 50 years old over this period of time, pointing to the rapid development of PFAS contamination and therefore the increasing potential risks [49].

Due to their high solubility and high persistence, PFAS compounds can migrate in air and water bodies leading to concentrated level in environment and therefore, pose high risk and toxicity on public health [49]. This increasing aquatic bioaccumulation, soil/groundwater uptake, fish, seafood, meat, and vegetables were identified as the most PFAS sources that human can uptake at various levels [31]. On the other hand, soil contamination is one of the main environmental impacts due to PFAS spread. Research suggests that PFAS risk is associated with a range of impacts on ecosystem services. Although the impact and the risk of PFAS still not fully understood due to the poor evidence of the linkage between human public health and PFAS levels, previous research has found the exposure to PFAS including PFOA may delay bone development and accelerated male puberty in mice [5]. However, the adverse health effects of PFOA and some other compounds have been confirmed by several researchers [8,31,110].

PFAS proved to have high resistant to temperature and bio accumulative [118]. Recently, research has found that humans are normally have a long half-life of serum elimination of PFOS, PFHS, and PFOA with the recent realization of considerable PFAS levels in various media including fish, birds, mammals and human blood seems to be alarming and call for urgent intervention to alleviate any further degradation in the public health as well as the ecosystem [17,18,118,119]. Niu et al. [119] found that PFAS compounds, including PFOS and PFOA, pose a high risk to human and public health. The high linkage between the PFOS and PFOA and neuropsychological development in children was investigated and realized by Niu et al. [119]. They found that the PFOS and PFOA increased the risk of development problem and had significant impact on human health including the personal-social skills particularly among females [7,119]. The recent discovery proved that PFOA has been linked to the increased incidence of weight loss and even a disturbance in lipid system when the tests were conducted on laboratory animals [118]. Moreover, recent research on animals suggested that PFAO and PFOS are among the main causes of cancer where the two compounds were classified as carcinogenic substances. The same effect on humans is suggested by the World health Organization (WHO) who found that both PFOA and PFOS are potential carcinogenic materials to human bodies [111]. Other studies found high linkage between PFOA exposure and high cholesterol leading to liver enzymes and kidney cancer [110]. Other studies have found increasing level of PFOS and PFOA in the blood samples of human population and wildlife reflecting that severity of the exposure to the widespread of PFAS chemicals [8]. Another good evidence of PFAS exposure and potential risk was reported by the ATSDR [8] where blood serum concentrations with high PFOS and PFOA level were found in workers living near potential PFAS facilities and industries compared with normal population [8].

In summary, despite the many trials to limit the PFAS spread and the endorsement of phasing out the main PFAS substances (expressed in PFOS and PFOA), other compounds including PFAAs and related substances are still widely used in various industries including fire-fighting foams, photographic, semiconductor and others [20]. Therefore, the detection of PFAS in human bodies have not decreased [20,31,32,49]. Research has found the level of adverse health impact with the level of significance are depending on the extent of exposure, the duration, and the persistence [10].

A study by the French total diet [112] found that mothers are likely to provide a pathway of PFAO to their children through breast feeding where PFOA was noticed in 77% of the breast milk samples at an average level of 0.041 ng/mL and a maximum level of 0.308 ng/mL [119]. Also, because of the their immature developing immune system and fast body growth, children are probably much more sensitive to the impacts of PFAS [8]. A more valid link between PFAO and the adverse human health was realized when a sample of around 69,000 people in the Mid-Ohio Valley were tested for PFAS as the analysis of the water supply system there revealed a considerable level of PFAO (>50 ng/L of PFOA) [5].

6. PFAS Water Quality Guidelines

6.1. Current Llegislations and Practices in Various Countries

PFAS guideline threshold values are affected by several factors, including social, political, and economic influences [5]. The variation in the regulatory values of PFAS across different guidelines can be easily addressed. One of the main reasons is the differences in toxicology decisions and differences in exposure parameters [4]. Moreover, PFAS compounds encounter both temporal and spatial variation and as emerging contaminants, the regulations are rapidly changing to account for the developing knowledge. However, the protection of the human health remains the main focus of the PFAS regulations and guidance across all regulations and standards [4]. The Interstate Technology and Regulatory Council (ITRC) indicated the significant variation in PFAS regulations by identifying the states that have different guideline for PFOA and/or PFOS levels in drinking water and groundwater from EPA's health advisories (HAs) [4]. One main reason for that is due to the different bodies that regulate the PFAS. Whilst the environmental perspectives of PFAS is regulated by the (EPA). Toxic Substances Control Act (TSCA), their use in food is regulated by FDA which is normally associated by lack of certain scientific evidence on their hazardous impact and exposure rate [120–122]. Despite the drinking water contamination is an ongoing major issue, it is somehow puzzling that until now there seem to be no federal PFAS drinking water standards in the USA. The absence of such federal PFAS regulations has led multiple US states to develop specific water guidelines which can support the decisions regarding the cleaning of the contaminated site as well as drinking water surveillance and treatment [5]. Until recently, no MCLs were established for PFAS chemicals although great efforts are being made towards initiating MCL for PFOA and PFOS by the EPA and other agencies.

This lack of evidence between the public heath adverse impacts and the PFAS level has resulted in undefined epidemiological evidence which in turns created considerable variation among the different water guidelines due to the uncertainties involved [5]. There is still uncertainty around the potential

PFAS risk to the human health due to the limited data on how people are exposed to the PFAS and for how long. The exposure level and the consequences of this exposure are also poorly understood [53].

The need for extensive research to alleviate this uncertainty and how the PFAS impact the health risk is still among the top priorities in order to expect a safe exposure for humans. Until now, the authors are aware that there is still urgent need to conduct more research in the areas of PFAS toxicity and exposure as this remains one of the knowledge gaps in PFAS field. This is obvious since the first US EPA preliminary drinking water health advisory level towards the negative impacts of PFOS and PFOA was released before PFAS became a public issue in 2015 upon the growing public trend towards limiting PFAS release. This was upon the realization of the PFAs toxicity. More recent guidelines referring to the PFAS as a significant toxic substance were published by USEPA in 2019. Unfortunately, PFAS legislations and regulations are challenged by the significant differences across PFAS compounds associated with the limited information that can be utilized to establish uniform legislations across the globe [110].

Another main concern is the influence of the manufacturing companies and the bias attitude that some researcher may have depending on the interest of the funding agencies. Cordner et al. [5] indicated that economic factors play vital role in directing the guideline levels where a case of litigation was revealed by Minnesota Attorney General against 3 M when the company used a scientific researcher to manipulate others research findings and undermine the health impact of PFAS in what was considered as a violation of scientific norms [5].

6.2. EPA-US Guidelines

USEPA has released non-regulatory concentrations of PFAS that addresses the PFAS health impact in reference to the exposure time. Since 2006, EPA has reviewed and regulated around 191 PFAS compounds through a combination of orders [110]. According to the EPA, the lifetime health advisory (LHA) of 70 ng/L for PFOA and PFOS in drinking water is set as guideline. This LHA is applicable to PFOA and PFOS individually while it is applicable to the sum of both compounds in the case of accidental High concentration as in the case of Australian standards [4]. This value seems to be less conservative compared with other global regulations. The global focus on the PFAS was developed rapidly since 2002 where more conservative levels were in place due to the growing concerns. More restrictions were placed on the "long chain" (C8) molecule PFOS as it was withdrawn from markets. However, the current regulation seem to be driven by many factors including financial factors, detection limit and many others [5,121]. In 2016, the EPA suggested its lifetime limit for PFOA and PFOS of 70 ng/L, individually or combined. However, some US states argued that EPA's guidelines are insufficient and does not address the potential associated health risk and hence various PFAS threshold values were developed different from the EPA ones. The state's water guideline levels for PFOA and/or PFOS ranged from 13 to 1000 ng/L compared with 70 ng/L by the EPA for both compounds individually and combined (Table 7). For example, Minnesota established state guideline levels that were lower than the EPA guidelines of 35 ng/L PFOA and 27 ng/L PFOS. On the other hand, New Jersey has proposed 14 ng/L MCLs for PFOA and 13 ng/L for PFOS, the first lowest guideline standard in the US [5].

In USA, MCLs for any PFAS have not been identified by EPA, though they recently declared their intention to "initiate steps to evaluate the need for a maximum contaminant level (MCL) for PFOA and PFOS". Identifying the MCL would increase the ability of EPA's authority to study further on PFAS pollution [119]. In May 2016, USEPA lowered its drinking water health standard to 0.07 µg/L for the two most frequently found PFAS, PFOA and PFOS. A lifetime drinking water health advisory (HA) for PFOA was issued by the USEPA on the basis of a reference dose (RFD) of 0.07 micrograms per liter (µg/L) based on a developmental toxicity analysis in mice [7]. The toxicity values of PFAS are site specific with the highly likely temporal and spatial variation of these values. Moreover, the rapidly developed analytical methods represents another challenge with the rapid changing regulations.

Some toxicity values are varying from one standard to another and there are no uniform standard PFAS toxicity values across all countries.

Table 7. US PFAO/PFOS drinking water guideline levels (After [5]).

Guideline	**Advisory Level**		**Reference Dose**	
	PFAO (ng/L)	**PFOS (ng/L)**	**PFAO (ng/kg-Day)**	**PFOS (ng/kg-Day)**
U.S. EPAa, 2016, Health Advisory Level	70	70	20	20
Alaska DECb, 2016, Groundwater cleanup level	400	400	20	20
Maine DEPb, 2016, Remedial action guideline	130	560	6	80
Minnesota DOH, 2017, Noncancer health-based level	35	27	18	5.1
New Jersey DEP, 2017, Maximum contaminant level	14	13	2	1.8
North Carolina DENRb, 2012, Interim maximum allowable concentration	1000	-	N/A	NA
Texas CEQb, 2017, Protective concentration level	290	560	15	20
Vermonta DEC/DOH,6 2016, Primary groundwater enforcement standard	20	20	20	20

EU guidelines just recently has initiated a preliminary guideline on maximum allowable PFAS concentrations. In Germany, a health-based guidance of maximum PFAS level was proposed by the drinking water commission under the Ministry of Health. The proposed value is based on the safe lifelong exposure for all population groups of 300 ng/L for both PFOA and PFOS. In Germany, upon the detection of PFOA in drinking water at concentrations up to 0.64 g/L, the German Drinking Water Commission (TWK) established the first health based lifelong PFAO and PFOS exposure of 0.3 g/L in drinking water in June 2006 [40]. Until recently, Italy, has no PFAS guidelines in drinking water and the PFAS regulations were introduced upon the extreme detection of PFAS in water bodies in an area of the Veneto Region [38]. The highest amount of PFAS in drinking water was enforced by the Italian National Health Institute to protect human's health risk with PFOS $\leq$ 30 ng/L, PFOA $\leq$ 500 ng/L, and other PFAS $\leq$500 ng/L. In Spain, frequent PFAS monitoring programs were carried out and water samples were regularly tested for various PFAS substance. PFAS level varied across Spain with the conclusion of an unlikely health risk under the detected PFOS and PFOA levels where the maximum average levels of PFOS and PFOA were 1.81 and 2.40 ng/L, respectively [39].

Canada has developed federal guidelines for a few PFAS levels to avoid any potential human health affect, while values to safeguard ecological receptors are offered for PFOS (Table 8) [118]. An in-depth study was carried out on ecotoxicology and toxicology, environmental fate and behavior, and exposure. In order to initiate toxicological reference values (TRVs), adequate information on the PFAS impact was obtained, while ECCC assessed the suitability of obtainable non- or low-effect ecotoxicological data for the derivation of PFOS recommendations for multiple matrices for the safety of different trophic levels. The degree of PFAS in drinking water, soil, groundwater, and bird eggs are now available in the Canadian PFAS regulation.

Table 8. Health based guidance for usage in field investigation in Canada [110].

PFAS Name	Acronym	Drinking Water Screening Value (ng/L)
perfluorobutanoate	PFBA	30
perfluorobutane sulfonate	PFBS	15
perfluorohexanesulfonate	PFHxS	0.6
perfluoropentanoate	PFPeA	0.2
perfluorohexanoate	PFHxA	0.2
perfluoroheptanoate	PFHpA	0.2
perfluorononanoate	PFNA	0.02
fluorotelomer sulfonate	6:2 FTS	0.2
fluorotelomer sulfonate	8:2 FTS	0.2

In Australia, PFASs have been widely used in several industrial applications. PFAS health-based guidance values for PFOS, PFOA and PFHxS, were developed by The Department of Health, Food Standards Australia (Table 9). The inconsistent release of PFAS in the environment with the multiple PFAS sources have created additional barrier in PFAS management. The knowledge gap regarding the PFAS spread in the Australian environment made the process of setting definite guidelines a bit complex. Gallen et al. [24], for example, found that the level of PFAS in in the treated WWTP effluent was higher than the wastewater influence.

Table 9. Health based guidance for utilization in site investigation in Australia [122].

Health Based Guideline Value	PFOS and PFHxS (ng)	PFOA (ng)	PFAO (ng)
Tolerable daily intake	20	160	0.16
Guideline for drinking water quality	70	560	0.56
Guideline value for Recreational water quality	2000	10,000	10

The guideline values considered the health of the general community due to the PFAS exposure due water and food consumption [122]. Through an intergovernmental agreement (IGA), Australia has been forced to agree on a National Structure for Reacting to PFAS contamination to restrict the spread of PFAS contamination due to increasing concerns about the risk of PFAS. Under the IGA, Australians have followed a policy to respond to PFAS pollution, introduced national environmental management of PFAS, and implemented guidelines to advise government agencies involved in responding to PFAS contamination. The guidance values based on health indicate the threshold value of the amount of PFAS in food or drinking water that an individual can consume without being affected over a lifetime.

7. Factors Contributing to Variation in PFAS Guideline Levels

Since the guidelines are always driven by the toxicology and the risk values identified by the human body, the regulations for PFA substances were viewed to limit the potential health impact and risk. Data obtained from toxicity tests in China showed that the criteria maximum concentration (CMC) for protection of aquatic organisms were 3.78 and 45.54 mg·L^{-1} for PFOS and PFAO, respectively which is higher than values derived in North America [20]. This variation indicates the challenges in setting out uniform PFAS guideline values due to the prevailing uncertainty in risk assessment and the lack of solid scientific background. Moreover, PFAS regulations are also influenced by the technical capacity and other socio-economic factors involved. There is a developing concern towards PFASs regulations due to the growing frequent detection of PFAS in drinking water in the US. The USEPA has issued a long-term health advisory PFAs level in drinking water of 70 ng/L (for combination of PFOS and/or PFOA). This regulation is intended to lower the number of individual PFASs reported. However, these guidelines of PFAS in drinking water have been dropped to 70 ng/L by many states, as in the case of New Jersey where a maximum PFAS level for PFOS was set at 13 ng/L and a 14 ng/L target for PFOA proposed [7]. Other states including California have adopted the same while the State of New York made it even lower with a maximum allowable PFAS value of 10 ng/L for PFOS and PFOA. Some other states went even for lower maximum allowable PFAS as in the case of New Hampshire and Michigan with an MCL of 12 ng/L for PFOA and 16 ng/L health-based value for PFO, respectively [7]. Much more conservative and lower levels of PFAS were proposed by Denmark, with a temporary level of 3 ng/L for PFOS being considered. The challenge in setting out these regulatory values remains the technical capacity in providing proper detection levels as well as the applicability of meeting these targets with the development of industrial activities and the lack of knowledge in identifying the toxicology of PFAs at these levels.

8. Conclusions

The intensive and the widespread nature of industries that are heavily releasing PFAS substances has contributed to a PFAS build up in the environment which presents a serious threat to human life. Occasional mysterious PFAS release, fate, transport, and exposure by many industries are adding more complexity for the policymakers. While the pathway of PFAS is still not very clear, PFAS characterization and behavior need to be better explained, particularly in terms of occurrence, transformation, and degradation pathway.

Although PFAS has been linked to various health risks, such as cancer, liver damage, and hormone disruption, the extent of this risk remains uncertain. This is due to the poor understanding of the risk scale resulting from the exposure frequency as well as the severity and the exposure duration. Since the PFAS in soil can leach down to great depths (around 15 m), PFOS levels in soil have been advised by governments including Australia to have a maximum of 20 ng/g for land uses and industry, which vary according to the land use. This leaching potential is getting more evident in case of point source pollution areas including landfill and treatment plants. However, the retention of PFOS increases with the clay content and the organic matter as well as the decrease of soil pH.

There is an ongoing effort by many governments to phase out PFAS substances and find alternatives to PFAS substances. This effort has caused PFAS levels to decline particularly in surface water. Whilst the alternative chemicals should be less toxic and not persist in the environment, the phasing out process has not resulted in PFAS elimination or decrease where many new industries are prone to PFAS release. This may require extra effort in understanding the measures to phase out.

It is concluded that the longer the exposure to PFAS compound, the higher the risk is due to the ubiquitous uptake of PFAS. PFAS high doses uptake. There is still a lack of precise knowledge around the PFAs toxicology and the threshold values at which PFAs can pose severe health risk. Moreover, this link is still lacking the interpretation of the relationship between the extent of the PFAS exposure and the associated impact, where no specific data can explain the impact of longevity, frequency, and severity of this impact. While the guidelines agree on considering human health as a base for any regulatory values, there seems to be a significant variation across the global guidelines in setting unified PFAS standards since the PFAS level and rate are inconsistent within the same country and across the globe. These discrepancies stem from the differences in PFAS sources, toxicology decisions, and exposure rates where PFAS compound can transform from one compound to another as well as transport with sediment and water far from the source based on the surrounding environment and therefore creating temporal and spatial variation. The review highlighted the need for further research towards identifying the characteristics, fate, frequency, and the severity of PFAS represented by the exposure time and extent to better understand the nature of PFAS pathways and exposure.

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References

1. Buck, R.C.; Franklin, J.; Berger, U.; Conder, J.M.; Cousins, I.T.; de Voogt, P.; Jensen, A.A.; Kannan, K.; Mabury, S.A.; van Leeuwen, S.P. Perfluoroalkyl and Polyfluoroalkyl Substances in the Environment: Terminology, Classification, and Origins. *Integr. Environ. Assess. Manag.* **2011**, *7*, 513–541. [CrossRef] [PubMed]
2. Fu, Y.; Wang, T.; Wang, P.; Fu, Q.; Lu, Y. Effects of age, gender and region on serum concentrations of perfluorinated compounds in general population of Henan, China. *Chemosphere* **2014**, *110*, 104–110. [CrossRef] [PubMed]
3. Clout, L.; Priddle, D.; Spafford, P.J. The Coalition against PFAS. 2018. Available online: https://cappfas.com/what-is-pfas/ (accessed on 19 December 2020).
4. ITRC. *PFAS Technichal and Regulatory Guidanace Document and Fact Sheet PFAS-1*; ITRC: Washington, DC, USA, 2020. [CrossRef]
5. Cordner, A.; de la Rosa, V.Y.; Schaider, L.A.; Rudel, R.A.; Richter, L.; Brown, P. Guideline levels for PFOA and PFOS in drinking water: The role of scientific uncertainty, risk assessment decisions, and social factors. *J. Expo. Sci. Environ. Epidemiol.* **2019**, *29*, 157–171. [CrossRef] [PubMed]
6. Giesy, J.P.; Kannan, K. Global Distribution of Perfluorooctane Sulfonate in Wildlife. *Environ. Sci. Technol.* **2001**, *35*, 1339–1342. [CrossRef] [PubMed]
7. USEPA. *Drinking Water Health Advisories for PFOA and PFOS.*; USEPA: Washington, DC, USA, 2016; pp. 1–4. Available online: https://www.epa.gov/sites/production/files/201605/documents/pfos_health_advisory_final-plain.pdf (accessed on 10 October 2020).
8. ATSDR. *Toxicological Profile for Perfluoroalkyls (Draft for Public Comment)*; Agency for Toxic Substances and Disease Registry: Atlanta, GA, USA, 2018. Available online: https://www.atsdr.cdc.gov/toxprofiles/tp.asp?id=1117&tid=237 (accessed on 28 October 2020).
9. Fàbrega, F.; Kumar, V.; Schuhmacher, M. PBPK modeling for PFOS and PFOA: Validation with human experimental data. *Toxicol. Lett.* **2014**, *230*, 244–251. [CrossRef]
10. Lake, N.; Cao, Y.; Cao, X.; Wang, H.; Wan, Y.; Wang, S. Assessment on the distribution and partitioning of perfluorinated compounds in the water and sediment. *Environ. Monit. Assess.* **2015**, *611*, 611. [CrossRef]
11. Zareitalabad, P.; Siemens, J.; Hamer, M.; Amelung, W. Perfluorooctanoic acid (PFOA) and perfluorooctanesulfonic acid (PFOS) in surface waters, sediments, soils and wastewater—A review on concentrations and distribution coefficients. *Chemosphere* **2013**, *91*, 725–732. [CrossRef]
12. Post, G.B.; Louis, J.B.; Cooper, K.R.; Boros-Russo, B.J.; Lippincott, R.L. Occurrence and Potential Significance of Perfluorooctanoic Acid (PFOA) Detected in New Jersey Public Drinking Water Systems. *Environ. Sci. Technol.* **2009**, *43*, 4547–4554. [CrossRef]
13. Jain, R.B. Time trends over 2003–2014 in the concentrations of selected perfluoroalkyl substances among US adults aged ≥ 20 years: Interpretational issues. *Sci. Total Environ.* **2018**, *645*, 946–957. [CrossRef]
14. Liu, B.; Zhang, H.; Li, J.; Dong, W. Perfluoroalkyl acids (PFAAs) in sediments from rivers of the Pearl River Delta, southern China. *Environ. Monit. Assess.* **2017**, *189*, 213. [CrossRef]
15. Dunn, A.M.; Hofmann, O.S.; Waters, B.; Witchel, E. Cloaking malware with the trusted platform module. In Proceedings of the 20th USENIX Security Symposium, San Francisco, CA, USA, 8–12 August 2011; pp. 395–410.
16. Vedagiri, U.K.; Anderson, R.H.; Loso, H.M.; Schwach, C.M. Ambient levels of PFOS and PFOA in multiple environmental media. *Remediat. J.* **2018**, *28*, 9–51. [CrossRef]
17. Goosey, E.; Harrad, S. Perfluoroalkyl compounds in dust from Asian, AustralianEuropean, and North American homes and UK cars, classrooms, and offices. *Environ. Int.* **2011**, *37*, 86–92. [CrossRef] [PubMed]
18. ITRC. *Environmental Fate and Transport for per-and Polyfluoroalkyl Substances*; ITRC: Washington, DC, USA, 2018; Available online: https://pfas-1.itrcweb.org/wp-content/uploads/2018/03/pfas_fact_sheet_fate_and_transport__3_16_18.pdf (accessed on 28 October 2020).
19. Kucharzyk, K.H.; Darlington, R.; Benotti, M.; Deeb, R.; Hawley, E. Novel treatment technologies for PFAS compounds: A critical review. *J. Environ. Manag.* **2017**, *204*, 757–764. [CrossRef] [PubMed]
20. Wang, T.; Wang, P.; Meng, J.; Liu, S.; Lu, Y.; Khim, J.S.; Giesy, J.P. A review of sources, multimedia distribution and health risks of perfluoroalkyl acids (PFAAs) in China. *Chemosphere* **2015**, *129*, 87–99. [CrossRef]

21. Page, D.; Vanderzalm, J.; Kumar, A.; Cheng, K.Y.; Kaksonen, A.H.; Simpson, S. Risks of perfluoroalkyl and polyfluoroalkyl substances (PFAS) for sustainable water recycling via aquifers. *Water* **2019**, *11*, 1737. [CrossRef]
22. Gallen, C.; Drage, D.; Eaglesham, G.; Grant, S.; Bowman, M.; Mueller, J.F. Australia-wide assessment of perfluoroalkyl substances (PFASs) in landfill leachates. *J. Hazard. Mater.* **2017**, *331*, 132–141. [CrossRef]
23. Cousins, I.T.; Vestergren, R.; Wang, Z.; Scheringer, M.; McLachlan, M.S. The precautionary principle and chemicals management: The example of perfluoroalkyl acids in groundwater. *Environ. Int.* **2016**, *94*, 331–340. [CrossRef]
24. Gallen, C.; Eaglesham, G.; Drage, D.; Nguyen, T.H.; Mueller, J.F. A mass estim Figure 2020 ate of perfluoroalkyl substance (PFAS) release from Australian wastewater treatment plants. *Chemosphere* **2018**, *208*, 975–983. [CrossRef]
25. Domingo, J.L.; Nadal, M. Human exposure to per-and polyfluoroalkyl substances (PFAS) through drinking water: A review of the recent scientific literature. *Environ. Res. J.* **2019**, *177*, 108648. [CrossRef]
26. Gonzalez, D.; Thompson, K.; Quiñones, O.; Dickenson, E.; Bott, C. Assessment of PFAS fate transport, and treatment inhibition associated with a simulated AFFF release within a WASTEWATER treatment plant. *Chemosphere* **2021**, *262*, 127900. [CrossRef]
27. Simon, J.A.; Cassidy, D.; Cherry, J.; Bryant, D.; Cox, D. PFAS Experts Symposium: Statements on regulatory policy, chemistry and analytics, toxicology, transport/fate, and remediation for per- and polyfluoroalkyl substances (PFAS) contamination issues. *Remediat. J.* **2019**, *29*, 31–48. [CrossRef]
28. Cerveny, D.; Grabic, R.; Fedorova, G.; Grabicova, K.; Turek, J.; Zlabek, V.; Randak, T. Fate of perfluoroalkyl substances within a small stream food web affected by sewage effluent. *Water Res.* **2018**, *134*, 226–233. [CrossRef] [PubMed]
29. Brusseau, M.L. Assessing the potential contributions of additional retention processes to PFAS retardation in the subsurface. *Sci. Total Environ.* **2018**, *613–614*, 176–185. [CrossRef] [PubMed]
30. Trudel, D.; Horowitz, L.; Wormuth, M.; Scheringer, M.; Cousins, I.T.; Hungerb, K. Estimating Consumer Exposure to PFOS and PFOA. *Risk Anal.* **2008**, *28*, 13–15. [CrossRef]
31. Sznajder-katarzy, K.; Surma, M.; Cie, I. A Review of Perfluoroalkyl Acids (PFAAs) in terms of Sources, Applications, Human Exposure, Dietary Intake, Toxicity, Legal Regulation, and Methods of Determination. *J. Chem.* **2019**, *2019*. [CrossRef]
32. Uebelacker, L.A. A Review of the Pathways of Human Exposure to Poly- and Perfluoroalkyl Substances (PFASs) and Present Understanding of Health Effects Elsie. *Physiol. Behav.* **2017**, *176*, 139–148. [CrossRef]
33. Pelch, K.E.; Reade, A.; Wol, T.A.M.; Kwiatkowski, C.F. Review article PFAS health e ff ects database: Protocol for a systematic evidence map. *Environ. Int. J.* **2019**, *130*, 104851. [CrossRef]
34. Zhao, L.; Zhu, L.; Zhao, S.; Ma, X. Sequestration and bioavailability of perfluoroalkyl acids (PFAAs) in soils: Implications for their underestimated risk. *Sci. Total Environ.* **2016**, *572*, 169–176. [CrossRef]
35. Yan, H.; Cousins, I.T.; Zhang, C.; Zhou, Q. Perfluoroalkyl acids in municipal landfill leachates from China: Occurrence, fate during leachate treatment and potential impact on groundwater. *Sci. Total Environ.* **2015**, *524–525*, 23–31. [CrossRef]
36. Eriksson, U.; Kärrman, A.; Rotander, A.; Mikkelsen, B.; Dam, M. Perfluoroalkyl substances (PFASs) in food and water from Faroe Islands, Environ. *Sci. Pollut. Res.* **2013**, *20*, 7940–7948. [CrossRef]
37. Lu, Z.; Lu, R.; Zheng, H.; Yan, J.; Song, L.; Wang, J.; Yang, H. Risk exposure assessment of per-and polyfluoroalkyl substances (PFASs) in drinking water and atmosphere in central eastern China. *Environ. Sci. Pollut. Res.* **2018**, *25*, 9311–9320. [CrossRef] [PubMed]
38. Mastrantonio, M.; Bai, E.; Uccelli, R.; Cordiano, V.; Screpanti, A.; Crosignani, P. Drinking water contamination from perfluoroalkyl substances (PFAS): An ecological mortality study in the Veneto Region, Italy. *Eur. J. Public Health* **2017**, *28*, 180–185. [CrossRef] [PubMed]
39. Domingo, J.L.; Ericson-jogsten, I.; Perello, G.; Nadal, M.; van Bavel, B.; Ka, A. Human Exposure to Perfluorinated Compounds in Catalonia, Spain: Contribution of Drinking Water and Fish and Shellfish. *Agric. Food Chem.* **2012**, *60*, 4408–4415. [CrossRef] [PubMed]
40. Wilhelm, M.; Bergmann, S.; Dieter, H.H. Occurrence of perfluorinated compounds (PFCs) in drinking water of North Rhine-Westphalia, Germany and new approach to assess drinking water contamination by shorter-chained C4–C7 PFCs. *Int. J. Hyg. Environ. Health* **2010**, *213*, 224–232. [CrossRef] [PubMed]

41. Schwanz, T.G.; Llorca, M.; Farré, M.; Barceló, D. Perfluoroalkyl substances assessment in drinking waters from Brazil, France and Spain. *Sci. Total Environ.* **2016**, *539*, 143–152. [CrossRef] [PubMed]
42. Quinete, N.; Wu, Q.; Zhang, T.; Yun, S.H.; Moreira, I.; Kannan, K. Specific profiles of perfluorinated compounds in surface and drinking waters and accumulation in mussels, fish, and dolphins from southeastern Brazil. *Chemosphere* **2009**, *77*, 863–869. [CrossRef]
43. Russell, M.H.; Berti, W.R.; Buck, R.C. Investigation of the Biodegradation Potential of a Fluoroacrylate Polymer Product in Aerobic Soils. *Environ. Sci. Technol.* **2008**, *42*, 800–807. [CrossRef]
44. Mak, Y.L.; Taniyasu, S.; Yeung, L.W.Y.; Lu, G.; Jin, L.; Yang, Y.; Lam, P.K.S.; Kannan, K.; Yamashita, N. Perfluorinated Compounds in Tap Water from China and Several Other Countries. *Environ. Sci. Technol.* **2009**, *43*, 4824–4829. [CrossRef]
45. Ghisi, R.; Manzetti, S. Accumulation of perfluorinated alkyl substances (PFAS) in agricultural plants: A review. *Environ. Res. J.* **2019**, *169*, 326–341. [CrossRef]
46. Bartolomé, M.; Gallego-picó, A.; Cutanda, F.; Huetos, O.; Esteban, M.; Pérez-Gómez, B.; Castaño, A. Perfluorinated alkyl substances in Spanish adults: Geographical distribution and determinants of exposure. *Sci. Total Environ.* **2017**, *603–604*, 352–360. [CrossRef]
47. Boiteux, V.; Dauchy, X.; Bach, C.; Colin, A.; Hemard, J.; Sagres, V.; Rosin, C.; Munoz, J. Science of the Total Environment Concentrations and patterns of perfluoroalkyl and polyfluoroalkyl substances in a river and three drinking water treatment plants near and far from a major production source. *Sci. Total Environ.* **2017**, *583*, 393–400. [CrossRef] [PubMed]
48. Loos, R.; Wollgast, J.; Huber, T. Polar herbicides, pharmaceutical products, perfluorooctanesulfonate (PFOS), perfluorooctanoate (PFOA), and nonylphenol and its carboxylates and ethoxylates in surface and tap waters around Lake Maggiore in Northern Italy. *Anal. Bioanal. Chem.* **2007**, *387*, 1469–1478. [CrossRef] [PubMed]
49. Lindstrom, A.B.; Strynar, M.J.; Libelo, E.L. Polyfluorinated compounds: Past, present, and future. *Environ. Sci. Technol.* **2011**, *45*, 7954–7961. [CrossRef] [PubMed]
50. Gellrich, V.; Brunn, H.; Stahl, T. Perfluoroalkyl and polyfluoroalkyl substances (PFASs) in mineral water and tap water. *J. Environ. Sci. Health Part A Toxic/Hazard. Subst. Environ. Eng.* **2013**, *48*, 129–135. [CrossRef] [PubMed]
51. Horst, J.; Mcdonough, J.; Ross, I.; Dickson, M.; Miles, J.; Hurst, J.; Storch, P. Advances in Remediation Solutions Water Treatment Technologies for PFAS: The Next Generation, Gr. *Water Monit. Remediat.* **2018**, *38*. [CrossRef]
52. Zhang, X.; Lohmann, R.; Dassuncao, C.; Hu, X.C.; Weber, A.K.; Vecitis, C.D.; Sunderland, E.M. Source Attribution of Poly- and Perfluoroalkyl Substances (PFASs) in Surface Waters from Rhode Island and the New York Metropolitan Area. *Environ. Sci. Technol. Lett.* **2016**, *3*, 316–321. [CrossRef]
53. Egeghy, P.P.; Judson, R.; Gangwal, S.; Mosher, S.; Smith, D.; Vail, J.; Cohen, E.A. The exposure data landscape for manufactured chemicals. *Sci. Total Environ.* **2012**, *414*, 159–166. [CrossRef]
54. Schultz, M.M.; Higgins, C.P.; Huset, C.A.; Luthy, R.G.; Barofsky, D.F.; Field, J.A. Fluorochemical mass flows in a municipal wastewater treatment facility. *Environ. Sci. Technol.* **2006**, *40*, 7350–7357. [CrossRef]
55. Yin, T.; Chen, H.; Reinhard, M.; Yi, X.; He, Y.; Gin, K.Y.H. Perfluoroalkyl and polyfluoroalkyl substances removal in a full-scale tropical constructed wetland system treating landfill leachate. *Water Res.* **2017**, *125*, 418–426. [CrossRef]
56. Bolan, N.; Sarkar, B.; Yan, Y.; Li, Q.; Wijesekara, H.; Kannan, K.; Tsang, D.C.W.; Schauerte, M.; Bosch, J.; Noll, H.; et al. Remediation of poly-and perfluoroalkyl substances (PFAS) contaminated soils–To mobilize or to immobilize or to degrade? *J. Hazard. Mater.* **2021**, *401*. [CrossRef]
57. Lee, H.; Tevlin, A.G.; Mabury, S.A.; Mabury, S.A. Fate of Polyfluoroalkyl Phosphate Diesters and Their Metabolites in Biosolids-Applied Soil: Biodegradation and Plant Uptake in Greenhouse and Field Experiments. *Environ. Sci. Technol.* **2014**, *48*. [CrossRef] [PubMed]
58. Karnjanapiboonwong, A.; Deb, S.K.; Subbiah, S.; Wang, D.; Anderson, T.A. Perfluoroalkylsulfonic and carboxylic acids in earthworms (Eisenia fetida): Accumulation and effects results from spiked soils at PFAS concentrations bracketing environmental relevance. *Chemosphere* **2018**, *199*, 168–173. [CrossRef] [PubMed]
59. Lechner, M.; Knapp, H. Carryover of Perfluorooctanoic Acid (PFOA) and Perfluorooctane Sulfonate (PFOS) from Soil to Plant and Distribution to the Different Plant Compartments Studied in Cultures of Carrots (*Daucus carota* ssp. *Sativus*), Potatoes (*Solanum tuberosum*), and cucumbers (*Cucumis sativus*). *J. Agric. Food Chem.* **2011**, 11011–11088. [CrossRef]

60. Stahl, T.; Heyn, J.; Thiele, H.; Hüther, J.; Failing, K.; Georgii, S.; Brunn, H. Carryover of Perfluorooctanoic Acid (PFOA) and Perfluorooctane Sulfonate (PFOS) from Soil to Plants. *Arch. Environ. Contam. Toxicol.* **2009**, *57*, 289–298. [CrossRef] [PubMed]
61. Cai, Y.; Chen, H.; Yuan, R.; Wang, F.; Chen, Z.; Zhou, B. Toxicity of perfluorinated compounds to soil microbial activity: Effect of carbon chain length, functional group and soil properties. *Sci. Total Environ.* **2019**, *690*, 1162–1169. [CrossRef] [PubMed]
62. Qiao, W.; Xie, Z.; Zhang, Y.; Liu, X.; Xie, S.; Huang, J.; Yu, L. Perfluoroalkyl substances (PFASs) influence the structure and function of soil bacterial community: Greenhouse experiment. *Sci. Total Environ.* **2018**, *642*, 1118–1126. [CrossRef]
63. Cai, Y.; Chen, H.; Yuan, R.; Wang, F.; Chen, Z.; Zhou, B. Metagenomic analysis of soil microbial community under PFOA and PFOS stress. *Environ. Res.* **2020**, *188*, 109838. [CrossRef]
64. Sun, Y.; Wang, T.; Peng, X.; Wang, P.; Lu, Y. Bacterial community compositions in sediment polluted by perfluoroalkyl acids (PFAAs) using Illumina high-throughput sequencing. *Environ. Sci. Pollut. Res.* **2016**, *23*, 10556–10565. [CrossRef]
65. Li, B.; Bao, Y.; Xu, Y.; Xie, S.; Huang, J. Vertical distribution of microbial communities in soils contaminated by chromium and perfluoroalkyl substances. *Sci. Total Environ.* **2017**, *599–600*, 156–164. [CrossRef]
66. Chen, H.; Wang, Q.; Cai, Y.; Yuan, R.; Wang, F.; Zhou, B.; Chen, Z. Effect of perfluorooctanoic acid on microbial activity in wheat soil under different fertilization conditions. *Environ. Pollut.* **2020**, *264*, 114784. [CrossRef]
67. Perkola, N.; Sainio, P. Survey of perfluorinated alkyl acids in Finnish effluents, storm water, landfill leachate and sludge. *Environ. Sci. Pollut. Res.* **2013**, *20*, 7979–7987. [CrossRef] [PubMed]
68. Busch, J.; Ahrens, L.; Sturm, R.; Ebinghaus, R. Polyfluoroalkyl compounds in landfill leachates. *Environ. Pollut.* **2010**, *158*, 1467–1471. [CrossRef] [PubMed]
69. Robey, N.M.; da Silva, B.F.; Annable, M.D.; Townsend, T.G.; Bowden, J.A. Concentrating Per- And Polyfluoroalkyl Substances (PFAS) in Municipal Solid Waste Landfill Leachate Using Foam Separation. *Environ. Sci. Technol.* **2020**, *54*, 12550–12559. [CrossRef] [PubMed]
70. Dauchy, X.; Boiteux, V.; Colin, A.; Hémard, J.; Bach, C.; Rosin, C.; Munoz, J.F. Deep seepage of per-and polyfluoroalkyl substances through the soil of a firefighter training site and subsequent groundwater contamination. *Chemosphere* **2019**, *214*, 729–737. [CrossRef]
71. Høisæter, Å.; Pfaff, A.; Breedveld, G.D. Leaching and transport of PFAS from aqueous film-forming foam (AFFF) in the unsaturated soil at a firefighting training facility under cold climatic conditions. *J. Contam. Hydrol.* **2019**, *222*, 112–122. [CrossRef]
72. Gao, Y.; Liang, Y.; Gao, K.; Wang, Y.; Wang, C.; Fu, J.; Wang, Y.; Jiang, G.; Jiang, Y. Levels, spatial distribution and isomer profiles of perfluoroalkyl acids in soil, groundwater and tap water around a manufactory in China. *Chemosphere* **2019**, *227*, 305–314. [CrossRef]
73. Wang, Y.; Fu, J.; Wang, T.; Liang, Y.; Pan, Y.; Cai, Y.; Jiang, G. Distribution of perfluorooctane sulfonate and other perfluorochemicals in the ambient environment around a manufacturing facility in china. *Environ. Sci. Technol.* **2010**, *44*, 8062–8067. [CrossRef]
74. Liu, B.; Zhang, H.; Yu, Y.; Xie, L.; Li, J.; Wang, X.; Dong, W. Perfluorinated Compounds (PFCs) in Soil of the Pearl River Delta, China: Spatial Distribution, Sources, and Ecological Risk Assessment. *Arch. Environ. Contam. Toxicol.* **2020**, *78*, 182–189. [CrossRef]
75. Chen, S.; Jiao, X.C.; Gai, N.; Li, X.J.; Wang, X.C.; Lu, G.H.; Piao, H.T.; Rao, Z.; Yang, Y.L. Perfluorinated compounds in soil, surface water, and groundwater from rural areas in eastern China. *Environ. Pollut.* **2016**, *211*, 124–131. [CrossRef]
76. Dalahmeh, S.; Tirgani, S.; Komakech, A.J.; Niwagaba, C.B.; Ahrens, L. Per-and polyfluoroalkyl substances (PFASs) in water, soil and plants in wetlands and agricultural areas in Kampala, Uganda. *Sci. Total Environ.* **2018**, *631–632*, 660–667. [CrossRef]
77. Armstrong, D.L.; Lozano, N.; Rice, C.P.; Ramirez, M.; Torrents, A. Temporal trends of perfluoroalkyl substances in limed biosolids from a large municipal water resource recovery facility. *J. Environ. Manag.* **2016**, *165*, 88–95. [CrossRef] [PubMed]
78. Herzke, D.; Olssonb, E.; Posner, S. Perfluoroalkyl and polyfluoroalkyl substances (PFASs) in consumer products in Norway—A pilot study. *Chemosphere* **2012**, *88*, 980–987. [CrossRef] [PubMed]
79. Clarke, B.O.; Anumol, T.; Barlaz, M.; Snyder, S.A. Investigating landfill leachate as a source of trace organic pollutants. *Chemosphere* **2015**, *127*, 269–275. [CrossRef] [PubMed]

80. Benskin, J.P.; Li, B.; Ikonomou, M.G.; Grace, J.R.; Li, L.Y. Per-and Polyfluoroalkyl Substances in Landfill Leachate: Patterns, Time Trends, and Sources. *Environ. Sci. Technol.* **2012**, *46*, 11532–11540. [CrossRef]
81. Eggen, T.; Moeder, M.; Arukwe, A. Municipal landfill leachates: A significant source for new and emerging pollutants. *Sci. Total Environ.* **2010**, *408*, 5147–5157. [CrossRef]
82. Fuertes, I.; Gómez-Lavín, S.; Elizalde, M.P.; Urtiaga, A. Perfluorinated alkyl substances (PFASs) in northern Spain municipal solid waste landfill leachates. *Chemosphere* **2017**, *168*, 399–407. [CrossRef]
83. Huset, C.A.; Barlaz, M.A.; Barofsky, D.F.; Field, J.A. Quantitative determination of fluorochemicals in municipal landfill leachates. *Chemosphere* **2011**, *82*, 1380–1386. [CrossRef]
84. Garg, S.; Kumar, P.; Mishra, V.; Guijt, R.; Singh, P.; Dumée, L.F.; Sharma, R.S. A review on the sources, occurrence and health risks of per-/poly-fluoroalkyl substances (PFAS) arising from the manufacture and disposal of electric and electronic products. *J. Water Process Eng.* **2020**, *38*, 101683. [CrossRef]
85. Loos, R.; Carvalho, R.; Anto, D.C.; Locoro, G.; Tavazzi, S.; Paracchini, B.; Ghiani, M.; Lettieri, T.; Blaha, L.; Jarosova, B.; et al. EU-wide monitoring survey on emerging polar organic contaminants in wastewater treatment plant effluents. *Water Res.* **2013**, *7*, 6475–6487. [CrossRef]
86. Zhang, Q.; Zhang, W.L.; Liang, Y.N. Adsorption of perfluoroalkyl and polyfluoroalkyl substances (PFASs) from aqueous solution—A review. *Sci. Total Sci. Total Environ.* **2020**, *748*, 142354. [CrossRef]
87. Xiao, F. Emerging poly-and perfluoroalkyl substances in the aquatic environment: A review of current literature. *Water Res.* **2017**, *124*, 482–495. [CrossRef] [PubMed]
88. Ochoa-Herrera, V.; Sierra-Alvarez, R. Removal of perfluorinated surfactants by sorption onto granular activated carbon, zeolite and sludge. *Chemosphere* **2008**, *72*, 1588–1593. [CrossRef]
89. Tang, H.; Xiang, Q.; Lei, M.; Yan, J.; Zhu, L.; Zou, J. Efficient degradation of perfluorooctanoic acid by UV–Fenton process. *Chem. Eng. J.* **2012**, *184*, 156–162. [CrossRef]
90. Gorenflo, A.; Veliizquez-Padon, D.; Frimmel, F.H. Nanofiltration of a German groundwater of high hardness and NOM content: Performance and costs. *Desalination* **2002**, *151*, 253–265. [CrossRef]
91. Hori, H.; Yamamoto, A.R.I. Efficient Decomposition of Environmentally Persistent Perfluorocarboxylic Acids by Use of Persulfate as a Photochemical Oxidant. *Environ. Sci. Technol.* **2005**, *39*, 2383–2388. [CrossRef] [PubMed]
92. Vecitis, C.D.; Wang, Y.; Cheng, J.I.E.; Park, H.; Mader, B.T. Sonochemical Degradation of Perfluorooctanesulfonate in Aqueous Film-Forming Foams. *Environ. Sci. Technol.* **2010**, *44*, 432–438. [CrossRef]
93. Franke, V.; McCleaf, P.; Lindegren, K.; Ahrens, L. Efficient removal of per-And polyfluoroalkyl substances (PFASs) in drinking water treatment: Nanofiltration combined with active carbon or anion exchange. *Environ. Sci. Water Res. Technol.* **2019**, *5*, 1836–1843. [CrossRef]
94. Zhang, D.; Luo, Q.; Gao, B.; Chiang, S.D.; Woodward, D.; Huang, Q. Sorption of perfluorooctanoic acid, perfluorooctane sulfonate and perfluoroheptanoic acid on granular activated carbon. *Chemosphere* **2016**, *144*, 2336–2342. [CrossRef]
95. Li, J.; Li, Q.; Li, L.; Xu, L. Removal of perfluorooctanoic acid from water with economical mesoporous melamine-formaldehyde resin microsphere. *Chem. Eng. J.* **2017**, *320*, 501–509. [CrossRef]
96. Du, Z.; Deng, S.; Chen, Y.; Wang, B.; Huang, J.; Wang, Y.; Yu, G. Removal of perfluorinated carboxylates from washing wastewater of perfluorooctanesulfonyl fluoride using activated carbons and resins. *J. Hazard. Mater.* **2015**, *286*, 136–143. [CrossRef]
97. Gao, Y.; Deng, S.; Du, Z.; Liu, K.; Yu, G. Adsorptive removal of emerging polyfluoroalky substances F-53B and PFOS by anion-exchange resin: A comparative study. *J. Hazard. Mater.* **2017**, *323*, 550–557. [CrossRef] [PubMed]
98. Franke, V.; Schäfers, D.; Lindberg, J.J.; Ahrens, L. Removal of per-and polyfluoroalkyl substances (PFASs) from tap water using heterogeneously catalyzed ozonation. *Environ. Sci. Water Res. Technol.* **2019**, *5*, 1887–1896. [CrossRef]
99. Andres, V.; Espana, A.; Mallavarapu, M.; Naidu, R. Treatment technologies for aqueous perfluorooctanesulfonate (PFOS) and perfluorooctanoate (PFOA): A critical review with an emphasis on field testing. *Environ. Technol. Innov.* **2015**, *4*, 168–181. [CrossRef]
100. Collings, A.F.; Gwan, P.B. Large scale environmental applications of high power ultrasound. *Ultrason. Sonochem.* **2010**, *17*, 1049–1053. [CrossRef] [PubMed]
101. Fernandez, N.A.; Rodriguez-Freire, L.; Keswani, M.; Sierra-Alvarez, R. Effect of chemical structure on the sonochemical degradation of perfluoroalkyl and polyfluoroalkyl substances (PFASs)†. *Environ. Sci. Water Res. Technol.* **2016**, 975–983. [CrossRef]

102. Zhao, L.; Bian, J.; Zhang, Y.; Zhu, L.; Liu, Z. Comparison of the sorption behaviors and mechanisms of perfluorosulfonates and perfluorocarboxylic acids on three kinds of clay minerals. *Chemosphere* **2014**, *114*, 51–58. [CrossRef]
103. Zhang, R.; Yan, W.; Jing, C. Mechanistic study of PFOS adsorption on kaolinite and montmorillonite, Colloids Surfaces A Physicochem. *Eng. Asp.* **2014**, *462*, 252–258. [CrossRef]
104. Wu, T.; Wu, Z.; Ma, D.; Xiang, W.; Zhang, J.; Liu, H.; Deng, Y.; Tan, S.; Cai, X. Fabrication of Few-Layered Porous Graphite for Removing Fluorosurfactant from Aqueous Solution. *Langmuir* **2018**, *34*, 15181–15188. [CrossRef]
105. Chen, X.; Xia, X.; Wang, X.; Qiao, J.; Chen, H. A comparative study on sorption of perfluorooctane sulfonate (PFOS) by chars, ash and carbon nanotubes. *Chemosphere* **2011**, *83*, 1313–1319. [CrossRef]
106. Zhang, Q.; Deng, S.; Yu, G.; Huang, J. Removal of perfluorooctane sulfonate from aqueous solution by crosslinked chitosan beads: Sorption kinetics and uptake mechanism. *Bioresour. Technol.* **2011**, *102*, 2265–2271. [CrossRef]
107. Fagbayigbo, B.O.; Opeolu, B.O.; Fatoki, O.S. Adsorption of perfluorooctanoic acid (PFOA) and perfluorooctane sulfonate (PFOS) from water using leaf biomass (Vitis vinifera) in a fixed-bed column study. *J. Environ. Health Sci. Eng.* **2020**, *18*, 221–233. [CrossRef] [PubMed]
108. Stebel, E.K.; Pike, K.A.; Nguyen, H.; Hartmann, H.A.; Klonowski, M.J.; Lawrence, M.G.; Collins, R.M.; Hefner, C.E.; Edmiston, P.L. Absorption of short-chain to long-chain perfluoroalkyl substances using swellable organically modified silica. *Environ. Sci. Water Res. Technol.* **2019**, *5*, 1854–1866. [CrossRef]
109. Qian, J.; Shen, M.; Wang, P.; Wang, C.; Hu, J.; Hou, J.; Ao, Y.; Zheng, H.; Li, K.; Liu, J. Co-adsorption of perfluorooctane sulfonate and phosphate on boehmite: Influence of temperature, phosphate initial concentration and pH. *Ecotoxicol. Environ. Saf.* **2017**, *137*, 71–77. [CrossRef] [PubMed]
110. Jian, J.M.; Zhang, C.; Wang, F.; Lu, X.W.; Wang, F.; Zeng, E.Y. Effect of solution chemistry and aggregation on adsorption of perfluorooctanesulphonate (PFOS) to nano-sized alumina. *Environ. Pollut.* **2019**, *251*, 425–433. [CrossRef] [PubMed]
111. Sunderland, E.M.; Hu, X.C.; Dassuncao, C.; Tokranov, A.K.; Wagner, C.C.; Allen, J.G. A review of the pathways of human exposure to poly-and perfluoroalkyl substances (PFASs) and present understanding of health effects. *J. Expo. Sci. Environ. Epidemiol.* **2019**, *29*, 131–147. [CrossRef] [PubMed]
112. Gebbink, W.A.; Berger, U.; Cousins, I.T. Estimating human exposure to PFOS isomers and PFCA homologues: The relative importance of direct and indirect (precursor) exposure Estimating human exposure to PFOS isomers and PFCA homologues: The relative importance of direct and indirect (precursor). *Environ. Int.* **2015**, *74*, 160–169. [CrossRef] [PubMed]
113. Skaar, J.S.; Ræder, E.M.; Lyche, J.L.; Ahrens, L.; Kallenborn, R. Elucidation of contamination sources for poly-and perfluoroalkyl substances (PFASs) on Svalbard (Norwegian Arctic). *Environ. Sci. Pollut. Res.* **2019**, *26*, 7356–7363. [CrossRef]
114. Andersen, M.E.; Butenhoff, J.L.; Chang, S.; Farrar, D.G.; Kennedy, G.L.; Lau, C.; Olsen, G.W.; Seed, J.; Wallace, K.B. Perfluoroalkyl Acids and Related Chemistries—Toxicokinetics and Modes of Action. *Toxicol. Sci.* **2008**, *102*, 3–14. [CrossRef]
115. Hoffman, K.; Webster, T.F.; Bartell, S.M.; Weisskopf, M.G.; Fletcher, T. Private Drinking Water Wells as a Source of Exposure to Perfluorooctanoic Acid Private Drinking Water Wells as a Source of Exposure to Perfluorooctanoic Acid (PFOA) in Communities Surrounding a Fluoropolymer Production Facility. *Environ. Health Perspect.* **2010**, *119*, 92–97. [CrossRef]
116. Longpré, D.; Lorusso, L.; Levicki, C.; Carrier, R.; Cureton, P. PFOS, PFOA, LC-PFCAS, and certain other PFAS: A focus on Canadian guidelines and guidance for contaminated sites management. *Environ. Technol. Innov.* **2020**, *18*, 100752. [CrossRef]
117. Seyoum, A.; Pradhan, A.; Jass, J.; Olsson, P. Perfluorinated alkyl substances impede growth, reproduction, lipid metabolism and lifespan in Daphnia magna. *Sci. Total Environ.* **2020**, *737*, 139682. [CrossRef] [PubMed]
118. Skogheim, T.S.; Villanger, G.D.; Weyde, K.V.F.; Engel, S.M.; Surén, P.; Øie, M.G.; Skogan, A.H.; Biele, G.; Zeiner, P.; Øvergaard, K.R.; et al. Prenatal exposure to perfluoroalkyl substances and associations with symptoms of attention-deficit/hyperactivity disorder and cognitive functions in preschool children. *Int. J. Hyg. Environ. Health* **2020**, *223*, 80–92. [CrossRef] [PubMed]

119. Niu, J.; Liang, H.; Tian, Y.; Yuan, W.; Xiao, H.; Hu, H.; Sun, X.; Song, X.; Wen, S.; Yang, L.; et al. Prenatal plasma concentrations of Perfluoroalkyl and polyfluoroalkyl substances and neuropsychological development in children at four years of age. *Environ. Health* **2019**, *18*, 53. [CrossRef] [PubMed]
120. McCarthy, C.; Kappleman, W.; DiGuiseppi, W. Ecological Considerations of Per-and Polyfluoroalkyl Substances (PFAS). *Curr. Pollut. Reports.* **2017**, *3*, 289–301. [CrossRef]
121. Cordner, A.; College, W.; Ave, B.; Walla, W.; States, U.; Richter, L.; Brown, P. Can Chemical Class Approaches Replace Chemical-by-Chemical Strategies? Lessons from Recent U.S. FDA Regulatory Action on per-and Polyfluoroalkyl Substances. *Environ. Sci. Technol.* **2016**, *50*, 12584–12591. [CrossRef] [PubMed]
122. Australian-Government. *Health Based Guidance Values for PFAS*; Australian-Government, Department of Health: Canberra, Australia, 2019.

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Perspective

Outside the Safe Operating Space of a New Planetary Boundary for Per- and Polyfluoroalkyl Substances (PFAS)

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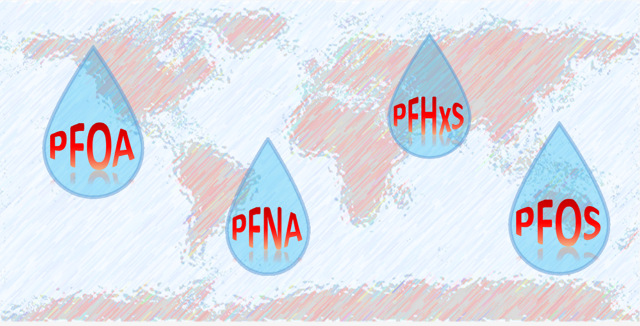

ABSTRACT: It is hypothesized that environmental contamination by per- and polyfluoroalkyl substances (PFAS) defines a separate planetary boundary and that this boundary has been exceeded. This hypothesis is tested by comparing the levels of four selected perfluoroalkyl acids (PFAAs) (i.e., perfluorooctanesulfonic acid (PFOS), perfluorooctanoic acid (PFOA), perfluorohexanesulfonic acid (PFHxS), and perfluorononanoic acid (PFNA)) in various global environmental media (i.e., rainwater, soils, and surface waters) with recently proposed guideline levels. On the basis of the four PFAAs considered, it is concluded that (1) levels of PFOA and PFOS in rainwater often greatly exceed US Environmental Protection Agency (EPA) Lifetime Drinking Water Health Advisory levels and the sum of the aforementioned four PFAAs (Σ4 PFAS) in rainwater is often above Danish drinking water limit values also based on Σ4 PFAS; (2) levels of PFOS in rainwater are often above Environmental Quality Standard for Inland European Union Surface Water; and (3) atmospheric deposition also leads to global soils being ubiquitously contaminated and to be often above proposed Dutch guideline values. It is, therefore, concluded that the global spread of these four PFAAs in the atmosphere has led to the planetary boundary for chemical pollution being exceeded. Levels of PFAAs in atmospheric deposition are especially poorly reversible because of the high persistence of PFAAs and their ability to continuously cycle in the hydrosphere, including on sea spray aerosols emitted from the oceans. Because of the poor reversibility of environmental exposure to PFAS and their associated effects, it is vitally important that PFAS uses and emissions are rapidly restricted.

KEYWORDS: *PFAS, planetary boundary, chemical pollution, environmental exposure*

■ INTRODUCTION

A recent review article in *Science*[1] highlighted the global threat posed by plastic pollution. These concerns were based on the high environmental persistence of plastics, the related "poor reversibility" and a range of potential effects. Other researchers, including ourselves,[2,3] have pointed out similar concerns related to highly persistent nonpolymeric substances, but these concerns are not equally obvious to the public compared to the concerns with plastics. The relatively high public concern regarding plastics is possibly driven by the visibility of plastic waste compared to nonpolymeric substances.[4] Clearly, both plastic pollution and pollution by highly persistent nonpolymeric substances lead to similar global problems. Persistence is generally seen as a less immediate hazardous property than toxicity, but it actually is the key factor that lets pollution problems spiral out of control.[2] This is because persistence enables chemicals to spread out over large distances, causes long-term, even life-long exposure, and leads to higher and higher levels in the environment as long as emissions continue. These increasing levels will with high probability sooner or later lead to adverse effects. Importantly, microplastic is under consideration for restriction in the EU because of the extreme persistence of plastics and the irreversibility of the exposure caused by plastic particles in the environment.[5]

Recently a group of scientists flagged the concerns regarding the inability of scientific analyses to keep pace with the amount of chemicals produced and released into the environment,[6] which limits the ability to discover new environmental threats in time. Others have similarly pointed out the need for precautionary chemicals managements; a notable example is the report, "Late Lessons from Early Warnings",[7] where many historical examples of global contamination problems are provided, often associated with persistent chemicals.

A well-known class of pollutants, the per- and polyfluoroalkyl substances (PFAS), have also recently featured in a review in *Science*.[8] The vast majority of PFAS are highly

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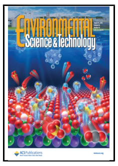

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persistent (based on the EU REACH definition whereby a substance is persistent if it is persistent itself or has persistent degradation products[9]), and this has been seen as basis for managing them as a chemical class.[3] While the review article in *Science* pointed out the ubiquity and high persistence of PFAS, it did not point out the current widespread and poorly reversible risks associated even with low-level PFAS exposures. It is hypothesized here that due to the global spread of PFAS, the irreversibility of exposure to PFAS, and the associated biological effects, a new planetary boundary for PFAS has been exceeded.

Unfortunately, although there are many thousands of substances defined as PFAS in use (PFAS include any substance with at least one $-CF_2-$ or $-CF_3$ moiety in its structure[10]), the current understanding of biological impacts is based primarily on studies of four perfluoroalkyl acids (PFAAs), namely, perfluorooctanesulfonic acid (PFOS), perfluorooctanoic acid (PFOA), perfluorohexanesulfonic acid (PFHxS), and perfluorononanoic acid (PFNA). Whereas all PFAS can be grouped into a class on the basis of their high persistence,[3] it is not possible to group many of them according to biological risk because of a paucity of data on exposure and effects for most PFAS.[11] Therefore, because of data gaps, the analysis presented here is based only on the four PFAAs mentioned above. In the following, we provide four pieces of evidence to support the claim that, even considering only these four PFAAs, the new planetary boundary for PFAS has been exceeded.

In the planetary boundary concept, an attempt is made to estimate the boundaries for "a safe operating space for humanity with respect to the functioning of the Earth System".[12,13] Chemical pollution was one of the original nine anthropogenic impacts for which planetary boundaries were postulated because it can influence Earth System functioning: "(i) through a global, ubiquitous impact on the physiological development and demography of humans and other organisms with ultimate impacts on ecosystem functioning and structure and (ii) by acting as a slow variable that affects other planetary boundaries."[12,13] The "chemical pollution" boundary was renamed as the "novel entities" (NEs) boundary by Steffen et al.,[14] where NEs are defined as "new substances, new forms of existing substances and modified life forms", including "chemicals and other new types of engineered materials or organisms not previously known to the Earth system as well as naturally occurring elements (for example, heavy metals) mobilized by anthropogenic activities". Several groups of scientists[6,15,16] have pointed out the challenges in quantifying the planetary boundary for NEs, and recently it was proposed to instead use various control variables to determine if the boundary is exceeded.[6] It is, in our opinion, an insurmountable task to quantify the boundary for all NEs because (1) there are critical data gaps for a large proportion of existing NEs, (2) NEs of various types and mixtures of NEs are continuously being generated and released to the environment, and (3) there are multiple possible effects (not only toxic effects) that individual NEs or groups/mixtures of NEs can cause. Several of the existing planetary boundaries are related to the release of NEs. For example, the boundaries for "stratospheric ozone depletion" and "climate change" address the release of ozone depleting substances and gases with global warming potential, respectively. Therefore, rather than being a single planetary boundary, the boundary for NEs can be thought of as a placeholder for multiple planetary boundaries for NEs that may emerge. It is argued here that PFAS define a new planetary boundary for NEs.

We argue that if drinking water health advisories and other guidelines designed to protect human health are exceeded due to the global environmental spread of PFAS, then there is a real danger of global health effects (e.g., affecting human physiology) occurring and that it can be argued that the planetary boundary for PFAS is exceeded. We do not deem it necessary to demonstrate the prevalence of global human health effects due to PFAS exposure to prove our hypothesis, and we hope that such widespread effects in the human population are never observed.

■ THE US EPA LIFETIME DRINKING WATER HEALTH ADVISORIES FOR PFOS AND PFOA ARE OFTEN LOWER THAN THEIR RESPECTIVE LEVELS IN RAINWATER AND THE DANISH DRINKING WATER LIMIT VALUE FOR Σ4 PFAS IS ALSO OFTEN LOWER THAN THE LEVEL OF Σ4 PFAS IN RAINWATER

In June 2022, the US Environmental Protection agency (EPA) announced the release of health advisories for four PFAS, including interim updated nonregulatory lifetime drinking water health advisories for PFOA and PFOS of 4 pg/L and 20 pg/L, respectively.[17] The US EPA health advisories identify the concentration of chemicals in drinking water at or below which adverse health effects are not anticipated to occur and, in divergence with previous advisories, are based on human epidemiology studies in populations exposed to these chemicals. The most sensitive noncancer effect and the basis for the risk assessment behind the interim updated health advisories for PFOA and PFOS is suppression of vaccine response (decreased serum antibody concentrations) in children. The US EPA's previous nonregulatory lifetime drinking water health advisories were 70 ng/L for the sum of concentrations of PFOS and PFOA. In 2020, the European Food Safety Authority (EFSA) published their Opinion on the risks to human health arising from the presence of PFAS in food[18] and proposed a group tolerable weekly intake (TWI) of 4.4 ng/kg body weight for the sum of PFOA, PFNA, PFHxS, and PFOS. On the basis of the available studies in animals and humans, effects on the immune system were considered the most critical for the basis of the risk assessment.[18] In June 2021, on the basis of the TWI in the EFSA Opinion, the Danish Environmental Protection Agency tightened their drinking water limit values and announced that drinking water must not contain more than 2 ng/L of Σ4 PFAAs.[19]

PFAS drinking water guidelines have progressively decreased over the last 22 years.[20] For example, in the US the PFOA drinking water guideline for West Virginia was 150 000 ng/L,[20] which is higher by a factor of 37.5 million than the recently announced US EPA drinking water lifetime advisory for PFOA of 4 pg/L. As a result of this decrease, international drinking water guidelines for PFAS are now close to, or even lower than, levels in precipitation. Humans residing in industrialized areas of the world do not often drink rainwater in modern life, but it should nevertheless be a reasonable expectation that the environment is clean enough that rainwater and mountain stream water fed by precipitation is safe to drink. Furthermore, in some parts of the world, notably in some arid and tropical regions, rainwater remains an important source of drinking water.[21]

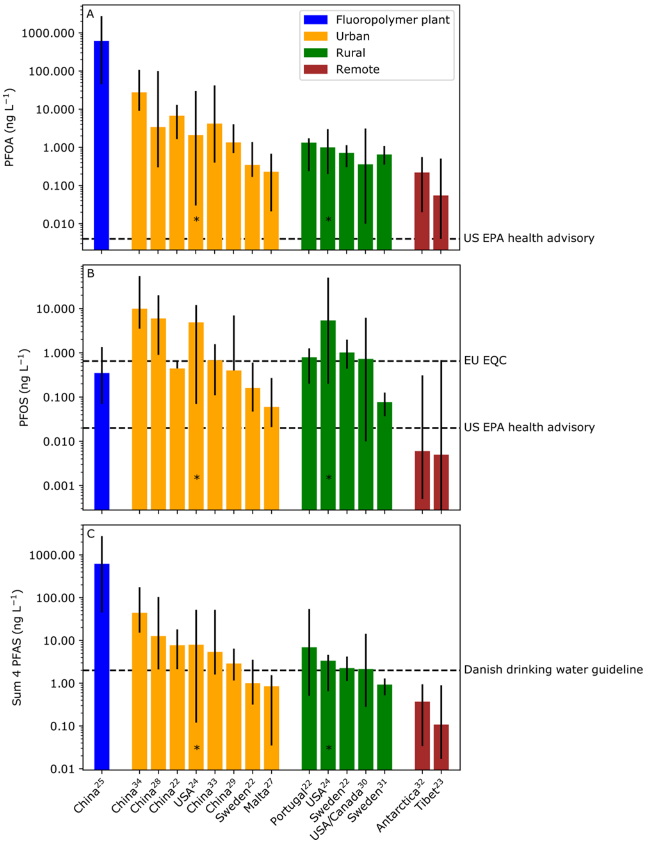

Figure 1. Levels of (A) PFOA, (B) PFOS, and (C) Σ4 PFAAs (PFOA + PFNA + PFHxS + PFOS) in wet deposition collected at various global locations from 2010 to the present. For one study,[24] it was not possible to derive median values and thus mean values are provided (indicated by *). The dashed line in (A) shows the US EPA health advisory for PFOA (0.004 ng/L), the dashed lines in (B) show the EU EQC for PFOS (i.e., 0.65 ng/L) and the US EPA health advisory for PFOS (i.e., 0.020 ng/L), and the dashed line in (C) shows the Danish drinking water guideline for Σ4 PFAAs (i.e., 2 ng/L). Bars indicate median values, and the uncertainty bars indicate minimum and maximum values. Wet deposition measurements for Σ4 PFAAs are ordered from high to low (from left to right) and sorted into four categories ("Fluoropolymer plant" indicates that samples were taken close to a fluoropolymer manufacturing plant; "Urban" indicates that samples were taken in cities or urbanized regions; "Rural" indicates that samples were taken in rural less-populated locations, and "Remote" indicates that samples were taken in regions with very low or nonexistent human populations). Some studies sampled wet deposition in multiple locations within one of the four categories, and thus data from these individual locations are grouped together in several bars. The raw data and a description of data treatment for figure preparation are provided in the Supporting Information.

In Figure 1, the levels of PFAS in precipitation are reviewed and compared to drinking water advisories for Denmark and the US EPA, which are the most stringent advisories known globally. The criteria for including/excluding studies for the selection shown in Figure 1 are (1) only studies which have precipitation samples are considered, (2) sampling and analysis was carried out after 2010, and (3) raw data or descriptive statistics (range and median or mean concentration) of the data were provided. Only data from 2010 or later were included because (1) these data are more recent and further from the 2000–2002 3M phase-out of long-chain PFAS chemistries and (2) there were large analytical improvements throughout the early 2000s as evidenced by the improvement in the fourth international interlaboratory study of 2011 compared to the three international interlaboratory studies conducted between 2004 and 2009.[22] Four precipitation studies were excluded because although the studies were published after 2010, the analysis was performed prior to 2010 (see Supporting Information).

In Figure 1A, the levels of PFOA in rainwater greatly exceed the US EPA drinking water health advisory for PFOA, even in remote areas (the lowest value for PFOA is for the Tibetan Plateau with a median of 55 pg/L,[23] which is approximately 14 times higher than the advisory). In Figure 1B, the levels of PFOS in rainwater are shown to often exceed the US EPA drinking water health advisory for PFOS, except for two studies conducted in remote regions (in Tibet and Antarctica). In Figure 1C, the levels of Σ4 PFAAs in precipitation are reviewed,[23−34] and it is shown that, in populated regions (defined as "urban" and "rural" in Figure 1), the levels would often exceed the Danish limit values for drinking water. In remote regions, with low human populations, the Σ4 PFAAs in rainwater also often exceeds the Danish drinking water limit value (Figure 1C). In Sweden, a national mapping of PFAS in municipal raw and drinking waters was undertaken in 2021.[35] About 49% of drinking waters in Sweden were found to contain ΣPFAS > 5 ng/L, and it was shown that the 4 PFAAs that are included in EFSA's risk assessment contributed a large fraction of the total PFAS measured. For comparison with the US, it was recently estimated[36] that at a concentration of 5 ng/L for combined PFOA + PFOS, 21−123 million people or 7−41% of the US population may have drinking water at or above that level, and at a concentration of 2.5 ng/L, it was estimated to be 76−205 million people or 25−68% of the population. The Swedish drinking water guideline for mitigation action (90 ng/L for Σ11 PFAS)[37] was previously based on the 2008 EFSA Scientific Opinion on PFOS and PFOA[38] and was recently reduced to 4 ng/L Σ4 PFAAs,[37] in light of the 2020 EFSA Opinion on PFAS.[18]

The US EPA health advisories seem not to be practically reachable without investment of huge cleanup costs in drinking water treatment plants given that most drinking water sources on the planet will have PFAS levels above the advisory levels. The US EPA health advisories are nonregulatory but demonstrating compliance to these guidelines would be an analytical challenge because modern methods are typically not able to achieve detection limits for PFOA below 4 pg/L in drinking water. Modern research laboratories have detection limits as low as 80 pg/L for PFOA and 100 pg/L for PFOS, respectively, in drinking water[39] and commercial laboratories tend to have much higher detection and quantification limits (e.g., Eurofins has quantification limits of about 1 ng/L for PFAS in water[40]). Achieving detection limits of <4 pg/L for PFOA in drinking water would be theoretically possible given that low pg/L levels have been previously measured in ocean water samples, even more than a decade ago.[41] Achieving such a low detection limit in drinking water would probably require extraction of larger than typical sample volumes and/or injection of larger extract volumes on the instrument. For example, the published method[39] that achieved 80 pg/L detection limits for PFOA was based on 10 mL samples and could be scaled to achieve <4 pg/L detection limits with larger sample volumes. It will also be important to have very low blank contamination because ultimately the blank levels and associated quality assurance will determine the detection limits that can be achieved.

THE EUROPEAN UNION (EU) ENVIRONMENTAL QUALITY STANDARD (EQS) FOR PFOS FOR FRESHWATERS IS OFTEN LOWER THAN LEVELS IN RAINWATER

In 2010, the National Institute for Public Health and the Environment (RIVM) in the Netherlands derived a risk-based maximum permissible concentration (MPC) for PFOS in freshwaters of 0.65 ng/L based on potential for secondary poisoning in humans due to fish consumption.[42] The MPC is a guideline level and it is defined as “the level at which no harmful effects are expected, based on annual average concentrations”. In 2013, PFOS and its derivatives were included in Directive 2013/39/EU and thus considered “Priority Hazardous Substances” under the Water Framework Directive (WFD) (2000/60/EC). Environmental Quality Standards (EQC) were, then, set for PFOS and its derivatives for freshwaters, marine waters, and biota. The EU annual average environmental quality standard (AA-EQS) for PFOS in Inland EU Surface Water was set at 0.65 ng/L, following the same reasoning used previously by RIVM. It is known that concentrations of PFOS in freshwaters regularly exceed the EQS,[43,44] but potentially of more concern is that the levels of PFOS in *rainwater* are equal to, or even exceed the EQS. As can be seen in Figure 1A), the levels of PFOS in rainwater in populated regions in the northern hemisphere in some cases exceed, or are close to, the EQC of 0.65 ng/L. Therefore, regardless of wastewater inputs to freshwaters, the EQC for PFOS will likely always be approached in populated regions, and often exceeded, as a result of the widespread presence of PFOS in atmospheric deposition.

Recently, authorities in the Stockholm metropolitan region have advised the public not to eat fish from lakes in the region.[45] This was not based on exceedance of the 0.65 ng/L EQS for PFOS and associated secondary poisoning but rather on exceedance of a temporary action level for fish of 9.1 ng/g PFOS set by the Swedish Food Agency.[37] The Swedish action level is considered temporary because it will be revised in the near future[37] according to the 2020 EFSA Scientific Opinion on the risks to human health arising from the presence of PFAS in food.[18] Given that the EU freshwater EQC is based on secondary consumption in humans because of fish consumption, there are grounds for revising the EQS based on the recent EFSA Opinion.[18] Such a revision of the freshwater EQS would likely result in a further reduction in its level and in basing the EQS on the sum of PFOA, PFNA, PFHxS, and PFOS.

THE DUTCH GUIDELINES FOR PFAS IN SOILS AND DREDGING MATERIAL WERE IMPOSSIBLE TO APPLY DUE TO THE UBIQUITY OF PFAS IN ATMOSPHERIC DEPOSITION

Recent guidelines set in July 2018 by the infrastructure ministry in the Netherlands stated that soil and dredging material should not contain concentrations of >0.1 μg/kg dry weight (dw) of either PFOS or PFOA.[46] As the levels of PFAS in soils often exceeded these guideline values, 70% of building projects involving soil removal and filling with excavated material were halted in the Netherlands.[47] Following builders’ protests, the Dutch government relaxed the guidelines.[48] Only a few studies have reported levels of PFAS in soils that have no known local PFAS source nearby. For example, Rankin et al. reported median PFOS and PFOA concentrations of 0.47 and 0.12 μg/kg dw for global soils,[49] whereas Sörengård et al. reported median PFOS and PFOA concentrations of 0.39 and 0.38 μg/kg dw in Swedish forest soils.[50] These reported soil levels illustrate the impossibility of complying with the Dutch guidelines before they were revised upward. The background soil contamination with PFAS is again a result of the environmental ubiquity of PFAAs in atmospheric deposition. If soils are amended with sewage sludge or biosolids, which is a common practice in agriculture in many countries, then soil levels will be further elevated and PFAS can leach to contaminate surface water and groundwater, including drinking water sources. On the basis of concerns regarding PFAS soil contamination, the US State of Maine passed a bill banning the use of biosolids in land applications unless, in the unlikely case, they could be shown to be PFAS free.[51]

THE CYCLING OF PFAAs IN THE WORLD’S HYDROSPHERE MEANS THAT LEVELS OF PFAAs IN RAINWATER WILL BE PRACTICALLY IRREVERSIBLE

Until recently, the common belief was that PFAAs would eventually wash off into the oceans where they would stay to be diluted over the time scale of decades.[52] A recent study,[53] however, has provided evidence that certain PFAS, notably the long-chain PFAAs, which include the 4 PFAAs included in EFSA’s TWI, can be significantly enriched on sea spray aerosols (SSA) and transported in the atmosphere back to shore where they will be deposited and contaminate freshwaters, drinking waters and surface soils.

This continual global cycling of PFAAs in the hydrosphere will lead to the continued exceedance of the above-mentioned guidelines. This finding is particularly worrying because (1) guideline values based on biological effects have continually decreased[20] and may not yet have reached the bottom as more scientific evidence emerges, (2) guidelines are currently based on only a few of the substances in the large PFAS class,[10] and (3) there is no evidence for the decline in environmental concentrations and thus environmentally derived exposures of PFAS.[54]

DISCUSSION

PFAS are a planetary boundary problem based on the criteria outlined by MacLeod et al.,[55] namely, (1) the diffuse PFAS pollution is global in its scale, (2) the effects are only now being discovered after the pollutants are already globally spread, and (3) now that the effects have been discovered they are poorly reversible or irreversible. As with most chemicals in use,[6] because of the lack of information, it is impossible to make a full assessment of the planetary boundary threat for the many thousands of PFAS in the class. Nevertheless, based on the four PFAAs considered here, it is concluded that in many areas inhabited by humans the planetary boundary for PFAS has been exceeded based on the levels in rainwater, surface water and soil, with all of these media being widely contaminated above recently proposed guideline levels. Although the global emissions of these 4 PFAAs have been reduced in recent years in most countries,[5256] these substances continue to remain in the environment due to their high persistence and will continually cycle in the hydrosphere.

The analysis presented here has purposefully referred to the most stringent PFAS guideline values on an international basis, which are not representative of international guideline values for PFAS. There is, for example, a large disagreement internationally, and even between individual states in the US,[20] regarding drinking water guidelines for PFAS. The various guidelines were developed by different scientists at different time points and the risk assessments are often based on varying end points. A clear and disturbing temporal trend emerges, however, with more recent guidelines being several orders of magnitude lower than older guidelines.[20] Guidelines in the US and Europe have been driven downward recently as a result of emerging evidence for the suppression of vaccine response in children.[57] We make no attempt to determine which of the many guidelines (see compilation[58]) is based on the strongest empirical evidence on effects because such a judgment is outside of our expertise. The point that we want to make is that the most stringent risk-based health advisories are often well below environmental levels, and this should be of concern and a reason for taking stringent measures.

Although PFAS are globally present in all environmental media and locations, there are still some few areas of the planet where the environmental levels of PFAS remain relatively low. However, even in these remote and sparsely populated regions, such as Antarctica and the Tibetan plateau, the most stringent PFAS guidelines are exceeded (Figure 1). These areas cannot support large populations and are not available for settlements where major parts of the population could move. In most other areas, PFAS guideline values are exceeded and this implies potential public health impacts: higher incidences (notably in large populations, i.e., many cases) of PFAS-related effects, such as reduced immune response, but also high additional costs for healthcare and, where possible, remediation.[59] Moreover, in many cases, PFAS-related impacts occur in combination with other environmental issues, such as water scarcity or pollution by other contaminants.

Finally, we conclude that PFAS define a new planetary boundary that has been exceeded, based on PFAS levels in environmental media being ubiquitously above guideline levels. Irrespective of whether or not one agrees with our conclusion that the planetary boundary for PFAS is exceeded, it is nevertheless highly problematic that everywhere on Earth where humans reside recently proposed health advisories cannot be achieved without large investment in advanced cleanup technology. Indeed, although PFOS and PFOA were phased out by one of the major manufacturers (3M) 20 years ago, it will take decades before levels in land-based water and precipitation approach low picogram per liter levels. Moreover, the problems associated with PFOS, PFOA, or Σ4 PFAAs are likely to be only the tip of the iceberg given that there are many thousands of PFAS in the class and the risks associated with most of them are unknown.[60] In view of the impacts of humanity's chemical footprint on planetary health, it is of great importance to avoid further escalation of the problem of large-scale and long-term environmental and human exposure to PFAS by rapidly restricting uses of PFAS wherever possible.[61] Furthermore, as has been stated by ourselves[3] and others[7] before, society should not continually repeat the same mistakes with other persistent chemicals.

■ ASSOCIATED CONTENT

Supporting Information

The Supporting Information is available free of charge at https://pubs.acs.org/doi/10.1021/acs.est.2c02765.

Raw data and references used to create Figure 1 (XLSX)

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Notes

The authors declare no competing financial interest.

Biography

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■ REFERENCES

(1) MacLeod, M.; Arp, H. P. H.; Tekman, M. B.; Jahnke, A. The Global Threat from Plastic Pollution. *Science* **2021**, *373* (6550), 61–65.

(2) Cousins, I. T.; Ng, C. A.; Wang, Z.; Scheringer, M. Why Is High Persistence Alone a Major Cause of Concern? *Environ. Sci.: Processes Impacts* **2019**, *21* (5), 781–792.

(3) Cousins, I. T.; DeWitt, J. C.; Glüge, J.; Goldenman, G.; Herzke, D.; Lohmann, R.; Ng, C. A.; Scheringer, M.; Wang, Z. The High Persistence of PFAS Is Sufficient for Their Management as a Chemical Class. *Environ. Sci.: Processes Impacts* **2020**, *22* (12), 2307–2312.

(4) Quantis. *Tackling the visible and invisible to close the plastic loop*. https://quantis-intl.com/ocean-plastics/ (accessed 2022-04-07).

(5) ECHA European Chemicals Agency. *Registry of restriction intentions until outcome—microplastics*. https://echa.europa.eu/sv/registry-of-restriction-intentions/-/dislist/details/0b0236e18244cd73 (accessed 2022-02-23).

(6) Persson, L.; Carney Almroth, B. M.; Collins, C. D.; Cornell, S.; de Wit, C. A.; Diamond, M. L.; Fantke, P.; Hassellöv, M.; MacLeod, M.; Ryberg, M. W.; Søgaard Jørgensen, P.; Villarrubia-Gómez, P.; Wang, Z.; Hauschild, M. Z. Outside the Safe Operating Space of the Planetary Boundary for Novel Entities. *Environ. Sci. Technol.* **2022**, *56* (3), 1510–1521.

(7) European Environment Agency (EEA). *Late Lessons from Early Warnings: Science, Precaution, Innovation*, Publication EEA Report No 1/2013; 2013. https://www.eea.europa.eu/publications/late-lessons-2.

(8) Evich, M. G.; Davis, M. J. B.; McCord, J. P.; Acrey, B.; Awkerman, J. A.; Knappe, D. R. U.; Lindstrom, A. B.; Speth, T. F.; Tebes-Stevens, C.; Strynar, M. J.; Wang, Z.; Weber, E. J.; Henderson, W. M.; Washington, J. W. Per- and Polyfluoroalkyl Substances in the Environment. *Science* **2022**, *375* (6580), No. eabg9065.

(9) ECHA European Chemicals Agency. *Management of PBT/vPvB substances under REACH-ECHA*. https://echa.europa.eu/management-of-pbt-vpvb-substances (accessed 2022-06-15).

(10) Wang, Z.; Buser, A. M.; Cousins, I. T.; Demattio, S.; Drost, W.; Johansson, O.; Ohno, K.; Patlewicz, G.; Richard, A. M.; Walker, G. W.; White, G. S.; Leinala, E. A New OECD Definition for Per- and Polyfluoroalkyl Substances. *Environ. Sci. Technol.* **2021**, *55* (23), 15575–15578.

(11) Cousins, I. T.; DeWitt, J. C.; Glüge, J.; Goldenman, G.; Herzke, D.; Lohmann, R.; Miller, M.; Ng, C. A.; Scheringer, M.; Vierke, L.; Wang, Z. Strategies for Grouping Per- and Polyfluoroalkyl Substances (PFAS) to Protect Human and Environmental Health. *Environ. Sci.: Processes Impacts* **2020**, *22* (7), 1444–1460.

(12) Rockström, J.; Steffen, W.; Noone, K.; Persson, Å.; Chapin, F. S.; Lambin, E. F.; Lenton, T. M.; Scheffer, M.; Folke, C.; Schellnhuber, H. J.; Nykvist, B.; de Wit, C. A.; Hughes, T.; van der Leeuw, S.; Rodhe, H.; Sörlin, S.; Snyder, P. K.; Costanza, R.; Svedin, U.; Falkenmark, M.; Karlberg, L.; Corell, R. W.; Fabry, V. J.; Hansen, J.; Walker, B.; Liverman, D.; Richardson, K.; Crutzen, P.; Foley, J. A. A Safe Operating Space for Humanity. *Nature* **2009**, *461* (7263), 472–475.

(13) Rockstrom, J.; Steffen, W.; Noone, K.; Persson, Å.; Chapin, F. S. I.; Lambin, E.; Lenton, T. M.; Scheffer, M.; Folke, C.; Schellnhuber, H. J.; Nykvist, B.; de Wit, C. A.; Hughes, T.; van der Leeuw, S.; Rodhe, H.; Sörlin, S.; Snyder, P. K.; Costanza, R.; Svedin, U.; Falkenmark, M.; Karlberg, L.; Corell, R. W.; Fabry, V. J.; Hansen, J.; Walker, B.; Liverman, D.; Richardson, K.; Crutzen, P.; Foley, J. Planetary Boundaries: Exploring the Safe Operating Space for Humanity. *Ecology and Society* **2009**, *14* (2), 1.

(14) Steffen, W.; Richardson, K.; Rockström, J.; Cornell, S. E.; Fetzer, I.; Bennett, E. M.; Biggs, R.; Carpenter, S. R.; de Vries, W.; de Wit, C. A.; Folke, C.; Gerten, D.; Heinke, J.; Mace, G. M.; Persson, L. M.; Ramanathan, V.; Reyers, B.; Sörlin, S. Planetary Boundaries: Guiding Human Development on a Changing Planet. *Science* **2015**, *347* (6223), 1259855.

(15) Persson, L. M.; Breitholtz, M.; Cousins, I. T.; de Wit, C. A.; MacLeod, M.; McLachlan, M. S. Confronting Unknown Planetary Boundary Threats from Chemical Pollution. *Environ. Sci. Technol.* **2013**, *47* (22), 12619–12622.

(16) Diamond, M. L.; de Wit, C. A.; Molander, S.; Scheringer, M.; Backhaus, T.; Lohmann, R.; Arvidsson, R.; Bergman, Å.; Hauschild, M.; Holoubek, I.; Persson, L.; Suzuki, N.; Vighi, M.; Zetzsch, C. Exploring the Planetary Boundary for Chemical Pollution. *Environment International* **2015**, *78*, 8–15.

(17) U.S. Environmental Protection Agency. *Lifetime Drinking Water Health Advisories for Four Perfluoroalkyl Substances (PFAS)*. https://www.epa.gov/system/files/documents/2022-06/prepublication-four-pfas-june-2022.pdf (accessed 2022-06-15).

(18) EFSA Panel on Contaminants in the Food Chain (EFSA CONTAM Panel). Risk to Human Health Related to the Presence of Perfluoroalkyl Substances in Food. *EFSA Journal* **2020**, *18* (9), No. e06223.

(19) Miljøministeriet. *Bekendtgørelse Om Vandkvalitet Og Tilsyn Med. Vandforsyningsanlæg*; 2021; Vol. BEK nr 2361 af 26/11/2021. https://www.retsinformation.dk/eli/lta/2021/2361.

(20) Post, G. B. Recent US State and Federal Drinking Water Guidelines for Per- and Polyfluoroalkyl Substances. *Environ. Toxicol. Chem.* **2021**, *40* (3), 550–563.

(21) Khayan, K.; Heru Husodo, A.; Astuti, I.; Sudarmadji, S.; Sugandawaty Djohan, T. Rainwater as a Source of Drinking Water: Health Impacts and Rainwater Treatment. *Journal of Environmental and Public Health* **2019**, *2019*, e1760950.

(22) Weiss, J. M.; van der Veen, I.; de Boer, J.; van Leeuwen, S. P. J.; Cofino, W.; Crum, S. Analytical Improvements Shown over Four Interlaboratory Studies of Perfluoroalkyl Substances in Environmental and Food Samples. *TrAC Trends in Analytical Chemistry* **2013**, *43*, 204–216.

(23) Chen, M.; Wang, C.; Gao, K.; Wang, X.; Fu, J.; Gong, P.; Wang, Y. Perfluoroalkyl Substances in Precipitation from the Tibetan Plateau during Monsoon Season: Concentrations, Source Regions and Mass Fluxes. *Chemosphere* **2021**, *282*, 131105.

(24) Pike, K. A.; Edmiston, P. L.; Morrison, J. J.; Faust, J. A. Correlation Analysis of Perfluoroalkyl Substances in Regional U.S. Precipitation Events. *Water Res.* **2021**, *190*, 116685.

(25) Liu, Z.; Lu, Y.; Shi, Y.; Wang, P.; Jones, K.; Sweetman, A. J.; Johnson, A. C.; Zhang, M.; Zhou, Y.; Lu, X.; Su, C.; Sarvajayakesavaluc, S.; Khan, K. Crop Bioaccumulation and Human

Exposure of Perfluoroalkyl Acids through Multi-Media Transport from a Mega Fluorochemical Industrial Park, China. *Environ. Int.* **2017**, *106*, 37−47.

(26) Shan, G.; Chen, X.; Zhu, L. Occurrence, Fluxes and Sources of Perfluoroalkyl Substances with Isomer Analysis in the Snow of Northern China. *J. Hazard. Mater.* **2015**, *299*, 639−646.

(27) Johansson, J. H.; Shi, Y.; Salter, M. E.; Cousins, I. T. Spatial Variation in the Atmospheric Deposition of Perfluoroalkyl Acids: Source Elucidation through Analysis of Isomer Patterns. *Environmental Science: Processes & Impacts* **2018**, *20* (7), 997−1006.

(28) Sammut, G.; Sinagra, E.; Helmus, R.; de Voogt, P. Perfluoroalkyl Substances in the Maltese Environment − (I) Surface Water and Rain Water. *Science of The Total Environment* **2017**, *589*, 182−190.

(29) Chen, H.; Zhang, L.; Li, M.; Yao, Y.; Zhao, Z.; Munoz, G.; Sun, H. Per- and Polyfluoroalkyl Substances (PFASs) in Precipitation from Mainland China: Contributions of Unknown Precursors and Short-Chain (C2C3) Perfluoroalkyl Carboxylic Acids. *Water Res.* **2019**, *153*, 169−177.

(30) Wang, S.; Lin, X.; Li, Q.; Li, Y.; Yamazaki, E.; Yamashita, N.; Wang, X. Particle Size Distribution, Wet Deposition and Scavenging Effect of per- and Polyfluoroalkyl Substances (PFASs) in the Atmosphere from a Subtropical City of China. *Science of The Total Environment* **2022**, *823*, 153528.

(31) Gewurtz, S. B.; Bradley, L. E.; Backus, S.; Dove, A.; McGoldrick, D.; Hung, H.; Dryfhout-Clark, H. Perfluoroalkyl Acids in Great Lakes Precipitation and Surface Water (2006−2018) Indicate Response to Phase-Outs, Regulatory Action, and Variability in Fate and Transport Processes. *Environ. Sci. Technol.* **2019**, *53* (15), 8543−8552.

(32) Filipovic, M.; Laudon, H.; McLachlan, M. S.; Berger, U. Mass Balance of Perfluorinated Alkyl Acids in a Pristine Boreal Catchment. *Environ. Sci. Technol.* **2015**, *49* (20), 12127−12135.

(33) Casas, G.; Martinez-Varela, A.; Vila-Costa, M.; Jiménez, B.; Dachs, J. Rain Amplification of Persistent Organic Pollutants. *Environ. Sci. Technol.* **2021**, *55* (19), 12961−12972.

(34) Han, T.; Gao, L.; Chen, J.; He, X.; Wang, B. Spatiotemporal Variations, Sources and Health Risk Assessment of Perfluoroalkyl Substances in a Temperate Bay Adjacent to Metropolis, North China. *Environ. Pollut.* **2020**, *265*, 115011.

(35) Lindfeldt, E.; Gyllenhammar, I.; Strandh, S.; Halldin Ankarberg, E. *Kartläggning av per- och polyfluorerade alkylsubstanser PFAS*, L-2021 nr 21; Livsmedelsverket: Uppsala, 2021; p 38. https://www.livsmedelsverket.se/globalassets/publikationsdatabas/rapporter/2021/l-2021-nr-21-kartlaggning-av-per-och-polyfluorerade-alkylsubstanser.pdf.

(36) Andrews, D. Q.; Naidenko, O. V. Population-Wide Exposure to Per- and Polyfluoroalkyl Substances from Drinking Water in the United States. *Environ. Sci. Technol. Lett.* **2020**, *7* (12), 931−936.

(37) Livsmedelsverket. *Riskhantering PFAS i dricksvatten och egenfångad fisk.* https://www.livsmedelsverket.se/foretagande-regler-kontroll/regler-for-livsmedelsforetag/dricksvattenproduktion/riskhantering-pfas-i-dricksvatten-egenfangad-fisk (accessed 2022-04-12).

(38) European Food Safety Authority (EFSA). Perfluorooctane Sulfonate (PFOS), Perfluorooctanoic Acid (PFOA) and Their Salts Scientific Opinion of the Panel on Contaminants in the Food Chain. *EFSA Journal* **2008**, *6* (7), 653.

(39) Skaggs, C. S.; Logue, B. A. Ultratrace Analysis of Per- and Polyfluoroalkyl Substances in Drinking Water Using Ice Concentration Linked with Extractive Stirrer and High Performance Liquid Chromatography − Tandem Mass Spectrometry. *Journal of Chromatography A* **2021**, *1659*, 462493.

(40) Eurofins. *Testing for PFOS, PFOA & GenX.* https://cdnmedia.eurofins.com/european-west/media/1926921/pfas_flier_oct_18_uk.pdf (accessed 2022-06-16).

(41) Yamashita, N.; Taniyasu, S.; Petrick, G.; Wei, S.; Gamo, T.; Lam, P. K. S.; Kannan, K. Perfluorinated Acids as Novel Chemical Tracers of Global Circulation of Ocean Waters. *Chemosphere* **2008**, *70* (7), 1247−1255.

(42) Moermond, C. T. A.; Verbruggen, E. M. J.; Smit, C. E. *Environmental Risk Limits for PFOS*, RIVM Report 601714013/2010; RIVM Rijksinstituut voor Volksgezondheid en Milieu, 2010; p 70. https://www.rivm.nl/bibliotheek/rapporten/601714013.pdf.

(43) Ahrens, L. Polyfluoroalkyl Compounds in the Aquatic Environment: A Review of Their Occurrence and Fate. *J. Environ. Monit* **2011**, *13* (1), 20−31.

(44) McLachlan, M. S.; Holmström, K. E.; Reth, M.; Berger, U. Riverine Discharge of Perfluorinated Carboxylates from the European Continent. *Environ. Sci. Technol.* **2007**, *41* (21), 7260−7265.

(45) Stockholms stad. *Avrådan från att äta insjöfisk.* https://via.tt.se/pressmeddelande/avradan-fran-att-ata-insjofisk?publisherId=1213538&releaseId=3302276 (accessed 2022-06-15).

(46) RIVM Rijksinstituut voor Volksgezondheid en Milieu. *Risicogrenzen voor PFOS, PFOA en GenX voor toepassen van grond en bagger*. https://www.rivm.nl/documenten/risicogrenzen-voor-pfos-pfoa-en-genx-voor-toepassen-van-grond-en-bagger (accessed 2022-02-23).

(47) RIVM Rijksinstituut voor Volksgezondheid en Milieu. *Temporary background values for PFAS in Dutch soil.* https://www.rivm.nl/en/news/temporary-background-values-for-pfas-in-dutch-soil (accessed 2022-02-23).

(48) RIVM Rijksinstituut voor Volksgezondheid en Milieu. *Tijdelijke landelijke achtergrondwaarden bodem voor PFOS en PFOA.* https://www.rivm.nl/documenten/tijdelijke-landelijke-achtergrondwaarden-bodem-voor-pfos-en-pfoa (accessed 2022-02-23).

(49) Rankin, K.; Mabury, S. A.; Jenkins, T. M.; Washington, J. W. A North American and Global Survey of Perfluoroalkyl Substances in Surface Soils: Distribution Patterns and Mode of Occurrence. *Chemosphere* **2016**, *161*, 333−341.

(50) Sörengård, M.; Kikuchi, J.; Wiberg, K.; Ahrens, L. Spatial Distribution and Load of Per- and Polyfluoroalkyl Substances (PFAS) in Background Soils in Sweden. *Chemosphere* **2022**, *295*, 133944.

(51) Maine State Legislature. *An Act To Prevent the Further Contamination of the Soils and Waters of the State with So-Called Forever Chemicals.* https://legislature.maine.gov/legis/bills/getPDF.asp?paper=HP1417&item=2&snum=130.

(52) Prevedouros, K.; Cousins, I. T.; Buck, R. C.; Korzeniowski, S. H. Sources, Fate and Transport of Perfluorocarboxylates. *Environ. Sci. Technol.* **2006**, *40* (1), 32−44.

(53) Sha, B.; Johansson, J. H.; Tunved, P.; Bohlin-Nizzetto, P.; Cousins, I. T.; Salter, M. E. Sea Spray Aerosol (SSA) as a Source of Perfluoroalkyl Acids (PFAAs) to the Atmosphere: Field Evidence from Long-Term Air Monitoring. *Environ. Sci. Technol.* **2022**, *56* (1), 228−238.

(54) Land, M.; de Wit, C. A.; Bignert, A.; Cousins, I. T.; Herzke, D.; Johansson, J. H.; Martin, J. W. What Is the Effect of Phasing out Long-Chain per- and Polyfluoroalkyl Substances on the Concentrations of Perfluoroalkyl Acids and Their Precursors in the Environment? A Systematic Review. *Environmental Evidence* **2018**, *7* (1), 4.

(55) MacLeod, M.; Breitholtz, M.; Cousins, I. T.; Wit, C. A. de; Persson, L. M.; Rudén, C.; McLachlan, M. S. Identifying Chemicals That Are Planetary Boundary Threats. *Environ. Sci. Technol.* **2014**, *48* (19), 11057−11063.

(56) Boucher, J. M.; Cousins, I. T.; Scheringer, M.; Hungerbühler, K.; Wang, Z. Toward a Comprehensive Global Emission Inventory of C4−C10 Perfluoroalkanesulfonic Acids (PFSAs) and Related Precursors: Focus on the Life Cycle of C6- and C10-Based Products. *Environ. Sci. Technol. Lett.* **2019**, *6* (1), 1−7.

(57) Grandjean, P.; Heilmann, C.; Weihe, P.; Nielsen, F.; Mogensen, U. B.; Timmermann, A.; Budtz-Jørgensen, E. Estimated Exposures to Perfluorinated Compounds in Infancy Predict Attenuated Vaccine Antibody Concentrations at Age 5-Years. *Journal of Immunotoxicology* **2017**, *14* (1), 188−195.

(58) Interstate Technology and Regulatory Council (ITRC). *PFAS—Per- and Polyfluoroalkyl Substances.* https://pfas-1.itrcweb.org/#1_7 (accessed 2022-06-15).

(59) Goldenman, G.; Fernandes, M.; Holland, M.; Tugran, T.; Nordin, A.; Schoumacher, C.; McNeill, A. *Cost of Inaction: A Socioeconomic Analysis of Environmental and Health Impacts Linked to Exposure to PFAS*; Nordisk Ministerråd, 2019. http://urn.kb.se/resolve?urn=urn:nbn:se:norden:org:diva-5514.

(60) Blum, A.; Balan, S. A.; Scheringer, M.; Trier, X.; Goldenman, G.; Cousins, I. T.; Diamond, M.; Fletcher, T.; Higgins, C.; Lindeman, A. E.; Peaslee, G.; de Voogt, P.; Wang, Z.; Weber, R. The Madrid Statement on Poly- and Perfluoroalkyl Substances (PFASs). *Environ. Health Perspect.* **2015**, *123* (5), A107−A111.

(61) Cousins, I. T.; Goldenman, G.; Herzke, D.; Lohmann, R.; Miller, M.; Ng, C. A.; Patton, S.; Scheringer, M.; Trier, X.; Vierke, L.; Wang, Z.; DeWitt, J. C. The Concept of Essential Use for Determining When Uses of PFASs Can Be Phased Out. *Environ. Sci.: Processes Impacts* **2019**, *21* (11), 1803−1815.

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Viewpoint

The True Cost of PFAS and the Benefits of Acting Now

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■ INTRODUCTION

Per- and polyfluoroalkyl substances (PFAS) are a class of over 9000 persistent hazardous chemicals used in industrial processes and consumer goods. They are ubiquitous in the environment and in people, who are exposed to PFAS via contaminated food and water, consumer products, and workplaces.[1] Exposure to several PFAS has been linked to a plethora of health effects in both animal and human studies, even at background levels. They are so environmentally persistent that they have been termed “forever chemicals.”

While in many ways PFAS contamination problems reflect broader issues with the chemicals regulatory system in the United States, a key feature of this industry is that only a handful of companies have produced the basic chemical building blocks for PFAS chemicals. These companies have known about the potential toxicity, human exposure, and extreme persistence of PFAS since the 1970s, yet have continued and expanded production.[2]

In the 2000s, in response to mounting pressure from the U.S. Environmental Protection Agency (EPA) about risks to human and environmental health, PFAS manufacturers agreed to phase out U.S. production of perfluorooctanoic acid (PFOA), perfluorooctanesulfonate (PFOS), and some related PFAS. Replacement PFAS, including new chemicals developed by industry, are widely used in more than 200 use categories,[3] despite growing concerns about exposures, persistence, and toxicity.[4]

The PFAS industry claims that the chemicals’ use in consumer goods and industrial applications brings wide benefits, valuing the U.S. fluoropolymer segment at $2 billion a year.[5] However, it fails to mention the costs of exposure, which are long-term, wide-ranging, routinely externalized onto

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the public, and disproportionately experienced. Focusing on a narrow, short-term view of PFAS benefits ignores how costs are displaced to communities and governments, despite existence of safer alternatives in most product sectors.

This review of the true costs of PFAS highlights the need to act now to ensure that exposures are capped at current levels by reducing the production and use of PFAS. It calls attention to systematic failures of U.S. chemical regulation, including inadequate premarket review of new compounds, data gaps that prevent and delay the regulation of existing chemicals, and the widespread externalization of social costs of pollution onto the public.

■ SNAPSHOT OF THE PROBLEM

Shifting the Burden to Public Utilities. Widespread contamination of surface water and groundwater due to industrial releases of PFAS or use of PFAS-containing firefighting foams is now a major problem in the United States and globally. An estimated 200 million U.S. residents, nearly two-thirds of the U.S. population, receive municipally provided drinking water that is contaminated with PFAS.[6]

Methods to reduce levels of PFAS in drinking water include filtration with granular activated charcoal treatment, reverse osmosis, ion exchange, or blending with less contaminated water from other sources, none of which fully eliminate PFAS. Municipalities may also opt to buy water from other distributors, but each method involves significant capital costs for new infrastructure and ongoing maintenance costs. For example, following extensive contamination by a PFAS manufacturer in the Cape Fear River watershed, Brunswick County, North Carolina spent $99 million on a reverse osmosis plant and will incur $2.9 million annually in operations expenses. Orange County, California estimates that the infrastructure needed to lower the levels of PFAS in its drinking water to the state's recommended levels will cost at least $1 billion.

These costs of cleaning up PFAS contamination of water are rarely internalized by chemical manufacturers or other responsible parties. Instead, they are usually displaced onto public utilities, their ratepayers, and state and local governments.

Communities with PFAS-contaminated drinking water also incur expenses related to testing and monitoring the contamination, informing the public, gathering information on treatment alternatives, studying the feasibility of infrastructure investments, and staff time for these projects. Low-income communities may be unable to cover such expenditures and often have few options for cost recovery, especially when the source of the PFAS contamination has not been determined. Additionally, PFAS contamination is likely to disproportionately impact vulnerable communities due to historic racial discrimination in housing and occupational sectors, and inequitable enforcement of environmental regulations that concentrate point sources of pollution proximal to these communities.

PFAS in wastewater can lead to additional expenses for public utilities. Wastewater treatment plants are designed to remove solids and pathogens, not persistent chemicals, and so any PFAS coming into the treatment plant are largely discharged into receiving waters or left as contaminants in sewage sludge. Needed treatment to remove contaminants will result in increased costs, and failure to treat may decrease existing revenue streams. For example, the public utility managing Merrimack, New Hampshire's wastewater currently earns $400,000 annually from processing sludge into compost for public sale as fertilizer. If the utility can no longer sell the sludge due to PFAS contamination, it will instead have to spend $2.4 million annually in landfill charges.

Other Externalized Costs of PFAS. Many other PFAS-related costs are routinely passed on to the public, rather than paid by the responsible polluters. For example, to prevent further contamination of water resources, the stock of fluorinated aqueous film-forming foams (AFFFs) still in place at military bases, airports, industrial sites, and local fire stations needs to be replaced with nonfluorinated foams. This requires collecting the AFFFs and then decontaminating or replacing equipment. The unused AFFFs and the PFAS-laden rinsewater must be contained, and no safe, permanent destruction methods currently exist.

The process of deciding what to do with hot spots of PFAS contamination is labor-intensive, time-consuming, and expensive. Testing of soil and water to determine the extent of contamination typically costs hundreds of dollars per sample, and few cleanup options exist. Landfilling of contaminated soil involves transportation costs and tip fees, and PFAS are only sequestered for the lifespan of the landfill. Incineration may destroy PFAS but only at extremely high temperatures, and has not been shown to work at large scale. Concerns about emissions from PFAS incineration, as well as public outrage at incineration testing in impacted communities, point to both health and political costs of PFAS incineration.

PFAS contamination may also reduce property values of homes and businesses. The discovery of water contamination, or even the perceived risk of potential contamination, can depress property values and stigmatize neighborhoods, potentially leading to lower home values and blocking residents' from selling properties, particularly when contamination achieves a level of public notoriety.[7]

Households and local businesses seeking to avoid exposure to contaminated drinking water may have to purchase bottled water or install and maintain home water filtration systems. In cases where the polluter is known, these costs may be recoverable through costly litigation. More often, however, the precise source of PFAS contamination is unclear, contested, or involves multiple polluters, making litigation or regulatory outcomes uncertain. Additionally, residents living outside of established boundaries or whose water is below specific action levels may not qualify for alternative water supplies, even if distribution systems exist.

Farms in areas with PFAS-contaminated water or soil may be forced to destroy harvests or products, or even to cease operation. As examples, dairy farms in more than one state were forced to dump milk contaminated with PFAS from agricultural applications of sludge and to euthanize their herds, while an organic farm near Colorado's Fort Peterson Air Force Base completely ceased production after learning that their irrigation water was highly contaminated.

Again, the governance and research expenses in such instances are substantial. In addition to technical expertise and staffing related to exposure assessment, human biomonitoring, and cleanup efforts, local and state governments must invest significant resources in public engagement and communications, and in managing PFAS programs and task forces. For example, North Carolina has allocated over $5 million for its PFAS Testing Network to address ongoing questions about PFAS exposure.

State and local governments may also incur significant legal expenses. States including New Hampshire and New Jersey have been sued by PFAS manufacturers opposed to health-protective drinking water regulations. States have occasionally received compensation from the companies responsible for PFAS pollution in their environs, including Minnesota ($850 million), Alabama ($39 million), and Michigan ($168 million).[8] The number of lawsuits and the size of settlements indicates the nation-wide scope of PFAS contamination and the costs of exposure. Legal actions such as these require significant time and resources from state-employed and contracted lawyers, consultants, and other professionals.

Moreover, these legal actions happen after the damage has occurred. Since complete remediation of PFAS in the environment is impossible at this time, exposures will remain for generations to come.

Health Impacts: The Biggest Externality. Exposure to PFAS via contaminated drinking water has been linked to kidney and testicular cancer, ulcerative colitis, pregnancy and fertility problems, liver diseases, thyroid disease, and high cholesterol.[1,9] PFAS exposure is also linked to immunotoxic effects, including decreased response to vaccines and possible increases in COVID-19 severity.[10] Even low-level exposure is associated with serious health consequences. For example, multiple studies have linked prenatal PFAS exposure with low birth weight, a particularly concerning end point that is associated with higher risk of cardiovascular disease, respiratory disease, and diabetes in adulthood, as well as impaired cognitive development and lower lifetime earnings.[11]

The impacts on human health due to PFAS exposure are immense. A recent analysis of impacts from PFAS exposure in Europe identified annual direct healthcare expenditures at €52−84 billion.[12] Equivalent health-related costs for the United States, accounting for population size and exchange rate differences, would be $37−59 billion annually. These costs are not paid by the polluter; they are borne by ordinary people, health care providers, and taxpayers.

Indirect social costs are also extensive, though more difficult to calculate. They include lost wages; lost years of life; reduced quality of life; increased stress, anxiety, and depression; and subsequent impacts on families and communities. Such social costs are quantifiable and can guide policy,[13] but no such analysis currently exists for health impacts from PFAS in the United States.

Finally, other significant health-related costs borne by government institutions and taxpayers include biomonitoring and health monitoring of exposed populations, and government research expenditures aimed at identifying PFAS toxicity and extent of exposures. In a more equitable world, this research would be carried out by the producer before the chemical came onto the market.

DISCUSSION

The health, societal, and economic impacts of contamination from PFAS production and use are multifaceted and broadly distributed. The costs of these impacts are long-term, incompletely understood, and externalized onto individuals, communities, and government at all levels, while profits accrue to corporations shielded from these costs by the protections built into our chemical regulatory laws and practices.[14] The continued use of PFAS will lead to increases in contamination and exposures in the future. But these exposures can be capped if steps are taken now to reduce and eventually phase out production and use of PFAS in all nonessential applications. In the meantime, the responsibility for paying for the legacy contamination should rest on the companies who continue to produce and market these chemicals even though they know about the chemicals' toxicity and extreme persistence.

Under a precautionary system of chemicals production in which companies had to demonstrate the safety of their products before accessing markets, costs could be substantially reduced by avoiding the production of toxic substances, and remaining costs would be internalized by PFAS producers into the price of their products. But in the United States, these costs are largely borne by the public and public institutions.

As this review of PFAS externalities shows, meaningful action must address not just remediation and cleanup of legacy contamination, but must also reduce current production and uses of PFAS, in order to limit the extent of future exposures. Class-based regulation of all PFAS is needed,[15] and California's recent action to regulate PFAS as a class in consumer products demonstrates that class-based restrictions are possible and desirable.[16]

Ubiquitous exposure to many toxic chemicals, not just PFAS, reflects a failure of regulatory systems to adequately reduce risk, and a privileging of short-term industry profits over long-term public health and environmental impacts. While the costs of drinking water treatment and PFAS remediation are substantial, the potential health-related costs of continued exposure to PFAS are much larger and will likely impact vulnerable communities disproportionately. Failing to take timely action to reduce the production and use of PFAS will result in exponentially higher costs to be paid by exposed populations for generations to come.

Understanding the true extent of these costs will clarify the benefits of improved regulatory controls and timely clean-ups. It will enable residents and policy makers to make informed decisions about who should rightfully bear responsibility for impacts and compensation. A strengthened regulatory system is needed, both in terms of enforcement of existing regulations and enactment of stronger, class-based laws to internalize the costs and reduce or eliminate the production of persistent, mobile, bioaccumulative, and toxic compounds. Only a strengthened regulatory system can adequately protect public health and the environment, and end the practice of forcing the public and future generations to bear the financial and health burden of pollution.

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■ REFERENCES

(1) ATSDR (Agency for Toxic Substances and Disease Registry). 2018. Toxicological Profile for Perfluoroalkyls (Draft for Public Comment). https://www.atsdr.cdc.gov/toxprofiles/tp200.pdf (accessed 2021/3/1).

(2) Richter, L.; Cordner, A.; Phil, B. Non-stick science: Sixty years of research and (in)action on fluorinated compounds. *Social Studies of Science* **2018**, *48*, 691−714.

(3) Glüge, J.; Scheringer, M.; Cousins, I. T.; DeWitt, J. C.; Goldenman, G.; Herzke, D.; Lohmann, R.; Ng, C. A.; Trier, X.; Zhanyun, W. An overview of the uses of per-and polyfluoroalkyl substances (PFAS). *Environ. Science: Process. Impacts* **2020**, *22*, 2345−2373.

(4) Wang, Z.; DeWitt, J. C.; Higgins, C. P.; Ian, T. C. A never-ending story of per-and polyfluoroalkyl substances (PFASs). *Environ. Sci. Technol.* **2017**, *51*, 2508−2518.

(5) Wood Environment & Infrastructure Solutions UK Limited. 2020. Socio-Economic Assessment of the US Fluoropolymer Industry -Executive Summary. https://fluoropolymerpartnership.com/wp-content/uploads/2020/03/Socio-Economic-Assessment-of-the-US-Fluoropolymer-Industry-Executive-Summary.pdf (accessed 2021/3/1).

(6) Andrews, D. Q.; Olga, V. N. Population-wide exposure to per-and polyfluoroalkyl substances from drinking water in the United States. *Environ. Sci. Technol. Lett.* **2020**, *7*, 931−936.

(7) Zabel, J. E.; Dennis, G. A hedonic analysis of the impact of LUST sites on house prices. *Resour. Energy Econ.* **2012**, *34*, 549−564.

(8) Gardella, John. 2020. Are the Floodgates Open for PFAS Product Liability Cases? *National Law Review XI* (94), https://www.natlawreview.com/article/pfas-product-liability-cases-are-floodgates-now-open (accessed 2021/3/1).

(9) C8 Science Panel. 2012. C8 Probable Link Reports. http://www.c8sciencepanel.org/prob_link.html (accessed 2021/3/1).

(10) Grandjean, P.; Timmermann, C. A.G.; Kruse, M.; Nielsen, F.; Vinholt, P. J.; Boding, L.; Heilmann, C.; Kåre, M. Severity of COVID-19 at elevated exposure to perfluorinated alkylates. *PLoS One* **2020**, *15*, No. e0244815.

(11) EFSA (European Food Safety Authority) Panel on Contaminants in the Food Chain. Risk to human health related to the presence of perfluoroalkyl substances in food. *EFSA J.* **2020**, *18*, 6223.

(12) Goldenman, G.; Menna, F.; Michael, H.; Tugce, T.; Amanda, N.; Cindy, S.; Alicia, M. *Cost of Inaction: A Socio-economic Analysis of Environmental and Health Impacts Linked to Exposure to PFAS*; Nordic Council of Ministers, Copenhagen, 2019. http://norden.diva-portal.org/smash/record.jsf?pid=diva2%3A1295959&dswid=6315 (accessed 2021/3/1).

(13) IOM (Institute of Medicine). *Cost of Environmental-Related Health Effects: A Plan for Continuing Study*; The National Academies Press: Washington, DC, 1981; DOI: 10.17226/812.

(14) Gold, S. C.; Wendy, E. W. Filling gaps in science exposes gaps in chemicals regulation. *Science* **2020**, *368*, 1066−1068.

(15) Kwiatkowski, C. F.; Andrews, D. Q.; Birnbaum, L. S.; Bruton, T. A.; DeWitte, J. C.; Knappe, D. R.U.; Maffini, M. V.; Miller, M. F.; Pelch, K. E.; Reade, A.; Soehl, A.; Trier, X.; Venier, M.; Wagner, C. C.; Wang, Z.; Arlene, B. Scientific basis for managing PFAS as a chemical class. *Environ. Sci. Technol. Lett.* **2020**, *7*, 532−543.

(16) Bălan, S. A.; Vivek, C. M.; Dennis, F. G.; André, M. A.. **2021**. Regulating PFAS as a chemical class under the California Safer Consumer Products Program. *Environ. Health Perspect. 129*, DOI: 10.1289/EHP7431.

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CONTROL AGENCY

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I hereby certify that this report was prepared by me or under my direct supervision and that I am a duly Licensed Professional Engineer under the laws of the state of Minnesota.

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Evaluation of Current Alternatives and Estimated Cost Curves for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water

May 2023

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Abbreviations

ADONA	8-dioxa-3H-perfluorononanoate
AFFF	aqueous film-forming foam
AIX	anion exchange
Barr	Barr Engineering Co.
BOD_5	five-day biochemical oxygen demand
CAA	Clean Air Act
$CaCO_3$	calcium carbonate
CCI	Construction Cost Index
CEPCI	Chemical Engineering Plant Cost Index
CO_2	carbon dioxide
COD	chemical oxygen demand
DOC	dissolved organic carbon
dtpd	dry tons of wastewater solids per day
EBCT	empty bed contact time
EIS	Environmental Impact Statement
ENR	Engineering News Record
FASA	perfluoroalkane sulfonamide
FTE	full-time employees
FTOH	fluorotelomer alcohol
FTS	fluorotelomer sulfonate
GAC	granular activated carbon
GHG	greenhouse gas
gpd	gallons per day
gpm	gallons per minute
HALT	high-temperature alkaline treatment
Hazen	Hazen and Sawyer
HBV	Health-Based Value
HF	hydrofluoric acid
HFPO-DA	propanoate
HRL	Health Risk Limit
HLR	hydraulic loading rate
HSDM	homogenous surface diffusion model
HWI	hazardous waste incinerator
ITRC	interstate technology regulatory council
kSCF	thousand standard cubic feet
lb	pound
MBR	membrane bioreactor
MC	modified clay
MF	microfiltration

mg/L	milligrams per liter
MGD	million gallons per day
min	minute
MPCA	Minnesota Pollution Control Agency
MSW	municipal solid waste
N-EtFOSA	N-ethyl perfluorooctane sulfonamide
N-EtFOSAA	N-ethyl perfluorooctane sulfonamido acetic acid
N-EtFOSE	N-ethyl perfluorooctane sulfonamidoethanol
NF	nanofiltration
ng/L	nanogram per liter
N-MeFOSAA	N-methyl perfluorooctanesulfonamido-acetic acid
NPDES	National Pollutant Discharge Elimination System
NPV	net present value
O&M	operations and maintenance
PASF	perfluoroalkane sulfonamido substance
PFAA	perfluoroalkyl acid
PFAS	per- and polyfluoroalkyl substances
PFBA	perfluorobutanoic acid
PFBS	perfluorobutane sulfonic acid
PFCA	perfluoroalkyl carboxylate
PFHpA	perfluoroheptanoate, perfluoroheptanoic acid
PFHxA	perfluorohexanoate, perfluorohexanoic acid
PFHxS	perfluorohexanesulfonic acid
PFOA	perfluorooctanoic acid
PFOS	perfluorooctane sulfonic acid
PFSA	perfluoroalkane sulfonate
PICs	products of incomplete combustion
Report	*Evaluation of Current Alternatives and Estimated Cost Curves for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water*
RO	reverse osmosis
SCWO	supercritical water oxidation
sq. ft.	square feet
SSI	sewage sludge incineration
SSOM	source-separated organic material
STU	standard treatment units
TDS	total dissolved solids
TOC	total organic carbon
TOP	total oxidizable precursors
TS	total solids
TSS	total suspended solids

UF	ultrafiltration
USD	United States Dollars
U.S. DOD	United States Department of Defense
U.S. EPA	United States Environmental Protection Agency
WRRF	water resource recovery facility

Executive Summary

This study develops alternatives to remove and destroy per- and polyfluoroalkyl substances (PFAS) from water resource recovery facility (WRRF) effluent, biosolids, mixed municipal solid waste (MSW) landfill leachate, and compost contact water (waste streams) using currently feasible technologies (i.e., could be built today). Barr Engineering Co. (Barr) and Hazen and Sawyer (Hazen) screened over 50 PFAS separation and destruction technologies for their ability to remove and destroy select PFAS to below current analytical reporting limits (a non-regulatory target established by the Minnesota Pollution Control Agency [MPCA] specifically for this study) and for their demonstrated commercial status. Thirteen technologies were retained for detailed consideration and assembled into alternatives, including destroying PFAS in final waste products. Assembled alternatives were ranked for criteria related to technical feasibility, economic feasibility, and byproducts management. Barr and Hazen retained two-to-four alternatives for each waste stream for preliminary design and cost estimating.

Currently, feasible technologies to separate PFAS from liquid waste streams are limited to sorption processes in pressure vessels (including granular activated carbon [GAC], anion exchange [AIX], and modified clay), reverse osmosis (RO) membrane separation, and foam fractionation. Feasible technologies to destroy PFAS from liquid media are currently limited to high-temperature incineration, thermal oxidation, and supercritical water oxidation (SCWO). Management of PFAS in biosolids remains a developing field with significant public and regulatory interest. Technologies selected as feasible at this time include SCWO, pyrolysis followed by thermal oxidation, and gasification followed by thermal oxidation.

Table ES-1 summarizes estimated capital and operations and maintenance (O&M) cost ranges for the two highest-ranking PFAS management alternatives for each waste stream for illustrative purposes. These estimates do not include pretreatment costs to achieve specified PFAS treatment process requirements. Pretreatment costs can, in some cases, be more expensive than PFAS removal and destruction. Requirements for both pretreatment and PFAS removal will vary significantly among sites and will need site-specific evaluations. Site-specific goals, conditions, and limitations may impact technology selection and implementation costs. Detailed PFAS removal cost estimates and cost curves for three facility sizes are included in this report. Based on our analyses, capital costs for removing PFAS from WRRF effluent and biosolids are similar, but O&M costs are significantly lower for biosolids treatment.

Table ES-1 Select capital and O&M cost ranges for highest-ranking alternatives

Waste Stream	Facility Size	Highest-Ranking Alternatives	Capital Cost Range (by facility)	Annual O&M Cost Range (by facility)	Relative Confidence in Ability to Reliably Meet PFAS Targets[2]
Municipal WRRF effluent	10 million gallons per day (MGD) (6,940 gpm) (similar to Mankato or Moorhead with a population of 45,000)	GAC with reactivation (Alt 1a)[1]	\$41M–\$88M	\$4.5M–\$9.6M	Medium-high (breakthrough of short-chain PFAS may limit reliability)
		GAC, single-use AIX with GAC reactivation and AIX high-temperature incineration (Alt 6a)[1]	\$80M–\$170M	\$6.1M–\$13M	High (two processes provide more controlled breakthrough)
Municipal WRRF biosolids	10 dry tons per day (estimated for 10 MGD WRRF)	SCWO[3]	\$40M–\$85M	\$0.47M–\$0.99M	Medium-high (limited testing at full-scale)
		Pyrolysis or gasification with thermal oxidation of pyrogas[1,3]	\$53M–\$110M	\$0.55M–\$1.2M	Medium-high high (limited testing at full scale)
Mixed MSW landfill leachate	0.014 MGD (10 gpm)	GAC with high-temperature incineration (Alt 1a)[1]	\$0.30M–\$0.60M	\$0.23M–\$0.48M	Medium (breakthrough of short-chain PFAS may limit reliability)
		Foam fractionation with high-temperature incineration of foamate (Alt 8a)	\$5.0M–\$11M	\$0.20M–\$0.42M	Low (limited removal of short-chain PFAS)
Compost contact water	0.014 MGD (10 gpm)	GAC with high-temperature incineration (Alt 1a)[1]	\$0.30M–\$0.60M	\$0.21M–\$0.44M	Medium (breakthrough of short-chain PFAS may limit reliability)
		Foam fractionation with high-temperature incineration of foamate (Alt 8a)	\$5.0M–\$11M	\$0.20M–\$0.42M	Low (limited removal of short-chain PFAS)

[1] Alternatives indicated likely need pretreatment processes to operate PFAS separation and destruction technologies. Pretreatment costs are not included in this table but are discussed in report sections for each waste stream.

[2] Relative ability to reliably meet PFAS targets reflects a combination of technology performance and reliability. For example, foam fractionation alternatives receive a “low” score because they are not expected to meet short-chain PFAS treatment targets. Alternately, single-process media filtration is expected to meet targets most of the time, except when a breakthrough event occurs. Hence, it receives a “medium” to “medium-high” score for reduced reliability. Breakthrough can be monitored and managed to limit PFAS reporting to effluent; however, targeting levels below analytical reporting limits for PFBA in high-concentration waste streams like landfill leachate could require media changeout every 2–4 weeks, which is on a similar time frame as analytical turnaround time for PFAS. Thus, PFAS breakthrough may not be detected in time for changeout, resulting in a lower reliability score for single-process media filtration for high PFAS concentration waste streams. Compared to single-process media filtration, dual-process media filtration receives a score of “high” because it is expected to allow for more time for monitoring breakthrough across four vessels instead of two and thus to more reliably meet PFAS targets.

[3] Biosolids costs are extrapolated from cost curves developed for this study.

Capital costs are driven by the recalcitrant and water-soluble nature of PFAS, which requires multiple additional processes, including pretreatment ahead of designated PFAS separation and destruction alternatives. Most currently available PFAS removal systems are modular, with limited economy-of-scale benefits for large facilities. O&M costs are driven by operational labor, energy use of high-temperature destruction technologies, and frequent sorption media changeout needed to achieve concentrations of short-chain PFAS below current method reporting limits (for alternatives with sorption media).

Costs were also evaluated with a lens on the cost per benefit provided by comparing the cost per mass of target PFAS removed between different waste streams and technologies over 20 years (detailed in Table 11-1). Treating wastewater biosolids or landfill leachate had the lowest cost per mass of target PFAS removed over 20 years (approximately \$0.7 million to \$4.0 million per pound of PFAS removed from biosolids and \$0.2 million to \$18 million per pound of PFAS removed from leachate). These costs are further described in Section 11.2. This cost range reflects the range of facility sizes analyzed here and the design basis influent PFAS concentrations established for this study.

When costs for individual facilities were extrapolated to the estimated number of WRRFs in Minnesota accepting greater than 0.05 MGD and mixed MSW landfills and composting sites, estimated costs for Minnesota could be at least \$14 billion for removing and destroying PFAS from WRRF effluent and biosolids, and at least \$105 million for removing and destroying PFAS from mixed MSW landfill leachate and compost contact water. These estimates, which include pretreatment, are summarized in Table ES-2 and further discussed in Section 11.3.

Table ES-2 Summary of estimated 20-year costs for managing PFAS in targeted waste streams in Minnesota[1]

Waste Stream	Estimated Number of Facilities	Range of Flows	Estimated 20-year costs for Minnesota (Millions of USD)[2]
Municipal WRRF effluent[3]	283	0.1–300 MGD	$12,000–$25,000
Municipal WRRF biosolids[4]	1 regional facility, plus 50 on-site facilities	50 dry tons of wastewater solids per day (dtpd) regional facility, on-site for 1–10 dtpd	$1,600–$3,300
Mixed MSW landfill leachate[5]	24	1–100 gpm	$77–$160
Compost contact water[6]	9	1–100 gpm	$28–$60

[1] This statewide evaluation carries additional uncertainty related to approximations for facility sizing, number of facilities, and degree of pretreatment needed. Costs are rounded to two significant figures. Costs are based on design basis concentrations selected to be typical of those reported in WRRF effluent (Helmer, Reeves, and Cassidy 2022; Coggan et al. 2019; Thompson et al. 2022), biosolids (Venkatesan and Halden 2013; Helmer, Reeves, and Cassidy 2022), landfill leachate (Lang et al. 2017), and compost contact water (Wood Environment & Infrastructure Solutions Inc. 2019).

[2] Twenty-year costs reflect net present value calculations using an interest rate of 7%.

[3] WRRF upgrade costs for effluent treatment are for PFAS separation and destruction using GAC adsorption with high-temperature incineration of media at flow rates below 1.1 MGD and GAC reactivation at higher flow rates. These include approximate costs for tertiary treatment retrofits (at WRRFs) or pretreatment processes (at landfill leachate and composting sites) likely needed at most facilities to provide the water quality required for GAC or RO feed. This analysis excludes WRRFs below 0.05 MGD.

[4] WRRF upgrade costs are for PFAS destruction in biosolids using pyrolysis or gasification with thermal oxidation of produced gasses. Costs include centrifuge dewatering to provide 25% solids material for process feed for each facility. These assume that WRRFs treating more than 0.1 MGD but producing less than 1 dtpd biosolids would ship to one regional, 50-dtpd pyrolysis facility. The costs shown here do not include transporting biosolids to that facility. These costs also do not include a pyrolysis/gasification facility with thermal oxidation for Minnesota's largest WRRF because costs for a facility of this size are not available.

[5] Costs are presented for 24 landfills, but the total number of landfills accepting mixed MSW in Minnesota is difficult to estimate due to mixed-use. Assumed equalization is present to limit peak leachate flows to twice the annual average leachate flow. Facility sizes are estimated based on publicly available data.

[6] Costs are presented for nine composting sites, but the total number of source-separated organic material (SSOM) composting sites is difficult to estimate due to mixed-use. Facility sizes are estimated based on publicly available data.

Most currently available PFAS destruction technologies are designed to treat concentrated waste streams rather than WRRF effluent water and are unlikely to be economically viable for most individual facilities. Regionalization of PFAS destruction may make financial sense for managing concentrated PFAS waste streams such as biosolids, foam fractionation foamate, GAC, and AIX resin. It may also be beneficial for treating high-concentration waste streams like landfill leachate, compost contact water, and biosolids from smaller facilities where on-site destruction is not economically viable. Evaluation of a regional high-temperature incineration facility for sorption media and a regional biosolids pyrolysis or gasification facility suggests that such facilities could potentially be economically viable when the fee structure is set appropriately to benefit the individual utilities and the regional facility. Other regionalization options that may become feasible include regional disposal of smaller volumes of foamate from foam fractionation using emerging destruction technologies such as SCWO, high-temperature alkaline treatment (HALT), or electrochemical oxidation.

Except for foam fractionation, liquid treatment technologies currently available at commercial scales are conventional water treatment technologies used in the water treatment industry for many years to treat other substances. While these technologies have been adapted at the commercial scale for PFAS treatment, many were not specifically designed for PFAS removal. New, targeted technologies to concentrate and destroy PFAS exist and have been demonstrated at bench- and pilot-scale. These newer technologies have the potential to reduce future capital and operating costs. However, these technologies are currently applied at small scales; for many of these newer technologies, performance and long-term maintenance needs have not been proven in full-scale implementations. In the future, these technologies may potentially be implemented at individual facilities rather than relying on regional or out-of-state high-temperature incineration facilities.

1 Introduction

Evaluation of Current Alternatives and Estimated Cost Curves for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water (Report) has been prepared for the MPCA by Barr, pursuant to a 2020 grant from the Legislative-Citizen Commission on Minnesota Resources. The objectives for this Report and the scope of work followed are summarized in this introductory section.

1.1 Background

Many studies and technologies investigated to-date have focused on the removal of PFAS from environmental media (Jin, Peydayesh, and Mezzenga 2021; Liu and Sun 2021; Berg et al. 2022). These often focus on waste streams with high concentrations of PFAS, including waste streams from primary and secondary manufacturers that produce or directly use PFAS-containing products and groundwater contaminated with aqueous film-forming foam (AFFF). The general management strategy for PFAS discharges has been industrial source-zone reduction and pretreatment.

While industrial sources reflect a significant input of PFAS to the environment, PFAS are also widely used in everyday consumer products and present in municipal wastewater, solid waste, and organic waste (Thompson et al. 2022; Roy et al. 2018; Hamid, Li, and Grace 2018; Choi et al. 2019). Thus, municipal waste streams act as a secondary avenue by which PFAS in consumer and industrial products enter the environment. Municipal WRRFs, in conjunction with municipal drinking water treatment and distribution, contribute to the recycling of PFAS in groundwater and surface drinking water supplies back into the environment.

Municipal and non-municipal solid and organic wastes are routed to landfills and commercial composting facilities, where PFAS in the wastes enter landfill leachate and compost contact water. Municipal WRRFs receive these streams along with municipal wastewater and industrial discharges. While these facilities are effective barriers for removing nutrients and organic matter, they are not designed or operated to remove PFAS, which report to both treated effluent (discharged to surface water) and the solid material removed from the wastewater. When these solids are stabilized through additional treatment, they are called "biosolids." Twenty-two percent of biosolids in Minnesota are applied to agricultural land for reuse of their nutrient value (from 171 WRRFs), 62% of biosolids are incinerated (from three WRRFs), and approximately 16% of biosolids are landfilled (from 27 WRRFs). This cycle is illustrated in Figure 1-1. This study targets the removal and destruction of PFAS from four waste streams, as illustrated by green circles.

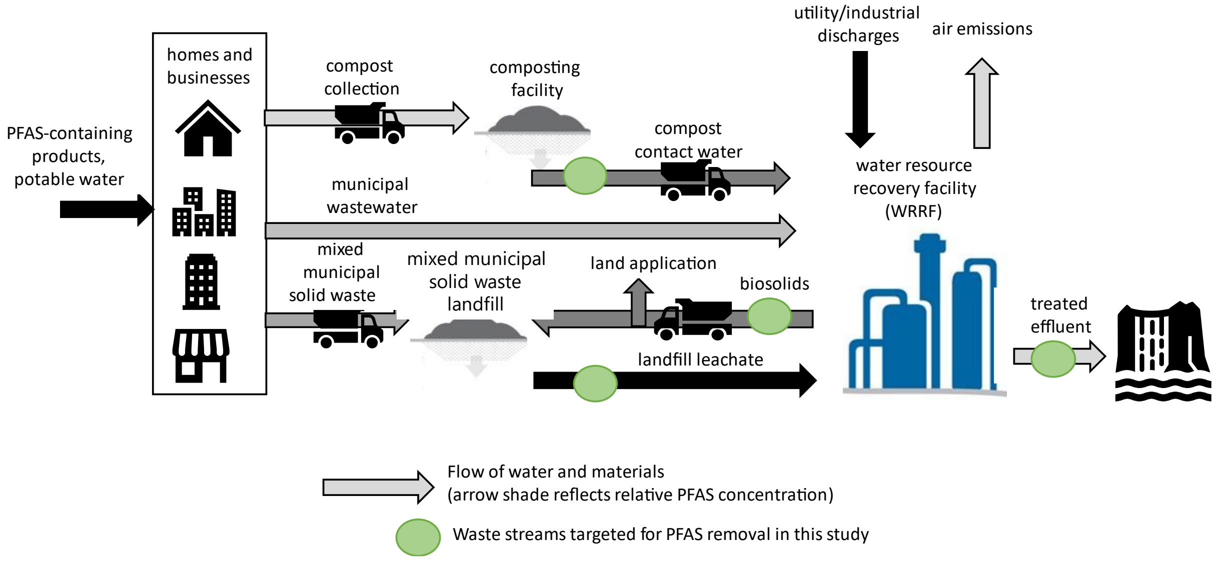


Figure 1-1 **PFAS use and disposal and target waste streams in this Report**

1.2 Study Objectives

The objective of this Report is to provide a conceptual understanding and costs for currently available approaches to **manage and destroy** per- and poly-fluoroalkyl substances (PFAS) down **to below current analytical reporting limits** in four waste streams:

- Municipal WRRF effluent
- Municipal WRRF biosolids
- Mixed MSW landfill leachate
- Compost contact water (also known as compost leachate)

Results may be used to inform future PFAS water quality standards and regulatory requirements associated with the four waste streams.

Manage and destroy: PFAS management has increasingly focused on the final fate of PFAS and the destruction (mineralization) of these chemicals to avoid potential re-release to the environment. This study aims to develop PFAS management strategies that mineralize PFAS to non-fluorinated end products. Management options that recycle PFAS to wastewater treatment plants, landfills, or other temporary reservoirs were not considered.

Below analytical reporting limits: PFAS guidelines for human and environmental health are rapidly evolving, and toxicity thresholds vary by compound. The MPCA set treatment targets for this study at current analytical reporting limits (5 ng/L for aqueous streams or 5 ng/g for biosolids) to reflect the target of complete PFAS removal (to the extent it can currently be measured) and destruction from both the targeted waste streams and other media that take on PFAS mass during treatment processes. Targeting analytical reporting limits for removal in this Report is aggressive, especially for short-chain PFAS with limited regulatory guidance. Many beneficial projects may target the mass removal of total PFAS or long-chain PFAS instead of a specific target concentration. However, evolving regulations around PFAS risk-based limits make it challenging to select alternative treatment targets.

Source-reduction strategies are not included as treatment options because the study evaluates end-of-pipe treatment options for PFAS. However, source-reduction strategies can potentially decrease source loads of PFAS routed to the waste streams under consideration, reducing the need to rely on end-of-pipe treatment. Source-reduction strategies may include phasing out PFAS use (Cousins et al. 2019) and continued treatment of concentrated point sources. As the cost of removing PFAS from the water phase is more related to the volume of water treated than to the mass of PFAS present, removal of PFAS from concentrated sources is more cost-effective than downstream treatment of lower-concentration WRRF flows.

The primary focus of the work completed for this Report was to use existing data sources to analyze the engineering feasibility and cost of current PFAS separation and destruction strategies to understand the management methods that could be employed at full-scale today to address PFAS in the waste streams considered. The alternatives evaluation did not include ongoing or future research and development activities to address PFAS. However, because PFAS treatment is a relatively new technical issue and a focus of significant current research, future treatment alternatives may significantly alter potential methods and costs for managing PFAS. A brief summary of ongoing and potential future research opportunities is discussed in Section 11.8.

The intent of this study is not to conduct a full life-cycle analysis of the treatment approaches identified. However, each treatment technology includes potential secondary effects (such as greenhouse gas emissions) associated with producing, transporting, reactivating, regenerating, and disposing of the treatment media and byproducts. These effects, which potentially negatively impact the sustainability of PFAS treatment, are discussed briefly in Section 11.4. It is recommended that facilities and sites seeking to implement PFAS management alternatives consider these sustainability effects and the full life-cycle impact of the technologies under consideration.

1.3 Use of Report

The purpose of this study is to provide a conceptual understanding and estimated costs for currently available approaches to **manage and destroy** per- and poly-fluoroalkyl substances (PFAS) in select waste streams outlined in this Report. Identification of possible treatment endpoints related to current PFAS analytical method detection limits and references to regulatory guidance/values existing at the time this study was completed are used to provide reference points and allow for cost calculations but are not intended to indicate the need for specific additional actions at specific facilities. That is beyond the scope of this study and should involve a wider consideration of each facility’s unique conditions and setting.

The capital and O&M cost estimates developed for this Report represent the best judgment of the experienced and qualified professionals familiar with this project. The estimated capital and O&M costs developed for preliminary PFAS management alternatives in this Report are intended to compare alternatives rather than represent estimates for specific projects. These cost estimates should not be directly applied in applications with different water or solids quality, operational constraints, or site-specific conditions. Barr has no control over the cost of labor, materials, equipment, or services furnished by others, any contractor's methods of determining prices, or competitive bidding or market conditions. Barr expects that costs for a specific site will vary from the opinions of probable capital and O&M costs for preliminary PFAS management alternatives developed for this Report. If managers of specific sites wish greater assurance as to probable costs, they should conduct a site-specific feasibility evaluation, including pilot testing, to characterize site-specific considerations and needs and to more accurately estimate PFAS management costs.

1.4 Report Structure and Study Scope

This Report is structured to address each of the four waste streams in separate sections to streamline use by readers primarily interested in one or two specific waste streams.

- Section 6 addresses municipal WRRF effluent.
- Section 7 addresses municipal WRRF biosolids.
- Section 8 addresses landfill leachate from mixed MSW landfills.
- Section 9 addresses compost contact water.

Key steps in the project and relevant sections for each are discussed below and summarized in Figure 1-2:

- **Selection of currently feasible PFAS separation and destruction technologies:** This task included reviewing and summarizing relevant peer-reviewed literature and vendor/owner data documenting the performance of currently available PFAS treatment and destruction systems (Section 3). Technology screening was used to evaluate the ability of individual technologies to separate or destroy PFAS. Technologies passing the screening phase were combined with others to develop a list of potential "treatment alternatives" for each waste stream.
- **Influent flow and load development for each waste stream**: Ranges of typical waste stream inflow rates per facility for each type of waste stream were developed to guide cost estimating. Typical water quality, biosolids characteristics, and PFAS content were also developed based on literature studies and MPCA-provided data (Sections 6.1, 7.1, 8.1, and 9.1). Many PFAS separation technologies require pretreatment to limit sorption media or membrane fouling, so pretreatment targets were also developed and used as assumed influent characteristics for those technologies. Example pretreatment trains with the potential to achieve influent water quality targets upstream were also developed (Sections 6.2, 7.2, 8.2, and 9.2); however, pretreatment requirements and costs are expected to vary significantly among facilities.
- **Assembly and evaluation of PFAS treatment alternatives:** Using currently feasible technologies, potentially viable treatment alternatives were assembled for each waste stream. These consist of multiple treatment technologies that, when employed together, could result in

complete or near-complete destruction of measurable PFAS (Section 4). Using previously developed influent flows and loads, assembled treatment alternatives were evaluated based on technical feasibility, economic feasibility, and byproducts management (Sections 6.2, 7.2, 8.2, and 9.2). The highest-ranking alternatives for each waste stream were carried forward to preliminary design and costing.

- **PFAS treatment design, costing, and cost curve development:** Preliminary designs were developed for selected PFAS treatment alternatives based on industry design standards and feedback from vendors. Vendor budgetary estimates and previous project costs were then used to develop cost curves for the influent flow ranges identified for each waste stream (within a margin of +50/-30%) (Sections 6.3, 7.3, 8.3, and 9.3).

- **Evaluation of regional PFAS destruction facility feasibility:** The feasibility, costs, and benefits of regionalized PFAS-destruction facilities (Section 10) were evaluated. Specific options evaluated in detail were a new, high-temperature incineration facility designed to handle sorption media from liquid treatment processes and a new pyrolysis facility designed to handle municipal wastewater biosolids from nearby WRRFs. Regional facility evaluations included consideration for economic viability, secondary impacts (externalities), and permitting and siting considerations.

Screen technologies	•Include broad list of separation and destruction technologies •Screen based on a.) commercial application and b.) PFAS separation/destruction efficacy
Develop influent flows and loads	•Set influent flow ranges per facility for each media •Develop influent characteristics and PFAS content for each media •Develop pretreatment targets, example pretreatment, and cost estimates
Assemble treatment alternatives	•Develop practical combinations of technologies passing screening •Include destruction to remove PFAS from the environment •Develop list of alternatives potentially applicable to specific media
Evaluate alternatives for each media	•Use subcriteria for technical and economic feasibility and byproducts management •Score each alternative for each subcriteria; weight and sum scores •Select alternatives for preliminary design and costing steps
Preliminary design and cost curve development	•Develop design basis for selected alternatives by media •Develop Class 5 capital and O&M cost estimates for systems of three sizes; synthesize into cost curves
Regional facility evaluation	•Develop design basis for two regional facility concepts: one for biosolids and one for liquid treatment residuals •Develop Class 5 capital and O&M cost estimates for one representative size •Conduct cost payback analysis under several operating conditions

Figure 1-2 Summary of steps used to develop treatment alternatives and cost analyses

1.5 Definitions

1.5.1 Waste Streams

Below are the definitions of the waste streams considered in this report.

- **Municipal wastewater or WRRF effluent:** Effluent from either conventional activated sludge WRRFs or stabilization pond WRRFs. Conventional activated sludge facilities were assumed to include primary treatment with physical screening, sedimentation, and/or floatation, followed by secondary treatment with mechanical aeration and clarification.

- **Municipal wastewater or WRRF biosolids:** For conventional activated sludge WRRFs, biosolids refer to a mixture of wastewater and solid materials separated from the wastewater during treatment (including microorganisms grown within the WRRF) and stabilized in an additional process, such as anaerobic digestion. For stabilization pond WRRFs, biosolids are the solids that accumulate and are naturally digested over time within the stabilization ponds.

- **Mixed MSW landfill leachate:** Liquid that drains from solid waste in a landfill, including water sourced from precipitation or originating in the landfilled material. Leachate available for treatment is assumed to be collected at a central location.

- **Compost contact water:** Water that has come into contact with source-separated organic material in the tipping area, material in the mixing area, rejects, residuals, or active compost at a commercial composting facility. The water originates from precipitation or moisture associated with compostable materials. Water originating from curing or finished storage areas at composting facilities is considered stormwater and managed separately from compost contact water.

1.5.2 Treatment and Destruction

Due to the evolving nature of PFAS regulation and treatment, semantics surrounding PFAS management can vary by application and location. The following definitions were developed for this study to support consistency and clarity:

- **Water treatment or biosolids treatment:** Technology or combination of technologies that alter the composition of water or biosolids to make it ready for another use.

- **PFAS removal:** The process of removing target PFAS from a specific waste stream of interest. This term can be applied to both separation and destruction technologies (Figure 1-3).

    - **PFAS separation:** Treatment technology or process that moves PFAS from one phase, waste stream, or media to another but does not destroy PFAS.

    - **PFAS destruction:** Treatment technology or process that destroys PFAS. For this Report, two categories of PFAS destruction were defined, as described below.

- **PFAS transformation or conversion:** The processes of transforming or converting PFAS to PFAS-reaction byproducts, which typically have a lower molecular weight and still include one or more carbon-fluorine bonds. These processes often apply to PFAS precursors converted or transformed into terminal perfluoroalkyl carboxylates (PFCAs).
- **PFAS mineralization:** The processes whereby PFAS are mineralized to carbon dioxide (CO_2), hydrofluoric acid (HF), and water. Defluorination, or the process of breaking the C-F bond, is a key step in PFAS mineralization processes because of the relatively high energy requirement. This definition is intended to be consistent with the United States Environmental Protection Agency's (U.S. EPA) definition of PFAS destruction as provided in the *Interim Guidance on the Destruction and Disposal of Perfluoroalkyl and Polyfluoroalkyl Substances and Materials Containing Perfluoroalkyl and Polyfluoroalkyl Substances* (U.S. EPA 2020).

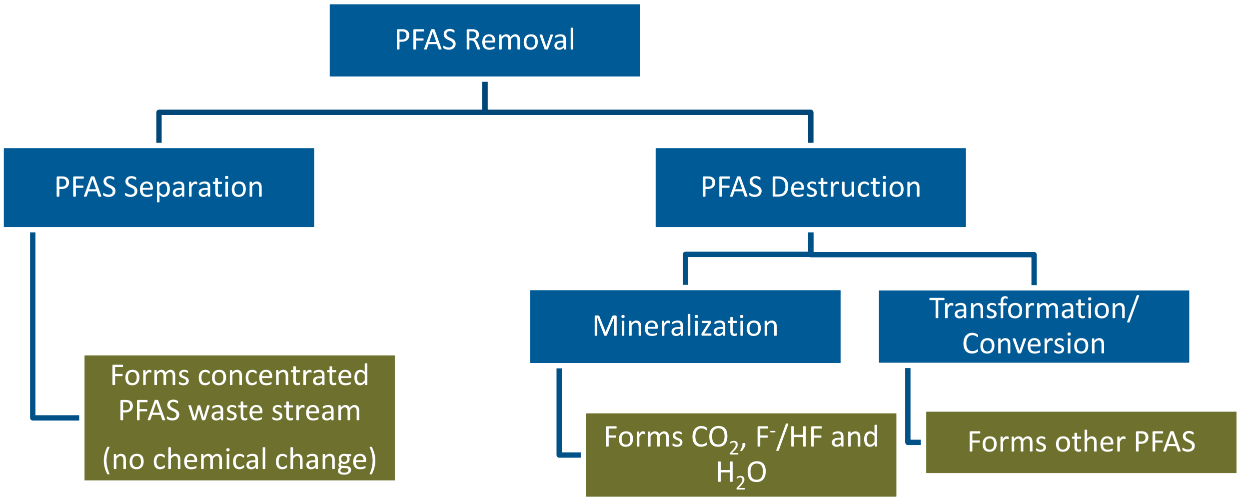


Figure 1-3 **Distinction between PFAS separation and PFAS destruction mechanisms**

2 PFAS Chemistry and Selection of PFAS Considered in this Study

2.1 PFAS Classes and Chemical Characteristics

Thousands of individual PFAS have been produced on a commercial scale, with newer chemicals replacing older ones as toxicology data and regulations evolve (U.S. EPA 2021a). Scientists are still learning about the chemical nature and health impacts of newer classes of PFAS (Evich et al. 2022). PFAS differ from each other based on various chemical characteristics, including carbon-chain length, the number of fluorinated carbon atoms, the inclusion of various functional groups, and per- versus polyfluorinated carbon chains (i.e., in perfluorinated compounds, all carbons are fully fluorinated while in polyfluorinated compounds, one or more carbon atom is not fully fluorinated). Major PFAS classifications for non-polymer PFAS are summarized below. These classifications are based on information from the *Naming Conventions Fact Sheet* from the Interstate Technology Regulatory Council (ITRC 2020).

- **Perfluoroalkyl acids (PFAAs):** This class includes perfluoroalkyl substances that have been most extensively studied, including perfluoroalkane carboxylates (PFCAs), such as perfluorooctanoic acid (PFOA) and perfluorobutanoic acid (PFBA), and perfluoroalkane sulfonates (PFSAs), such as perfluorooctane sulfonic acid (PFOS) and perfluorobutane sulfonic acid (PFBS).

- **Perfluoroalkane sulfonamides (FASAs):** This class includes perfluoroalkyl substances used as surfactants and in surface protection products. They can also be intermediate environmental transformation products. Representative compounds include perfluorooctane sulfonamide (PFOSA or FOSA) and N-ethyl perfluorooctane sulfonamide (N-EtFOSA).

- **Fluorotelomer substances:** This class includes polyfluorinated substances created via fluorotelomerization and used as major raw materials. This class also includes fluorotelomers, environmental transformation products and intermediates; they are also potential PFCA precursors. Representative compounds include 8:2 fluorotelomer sulfonate (8:2 FTS) and 10:2 fluorotelomer alcohol (10:2 FTOH).

- **Perfluoroalkane sulfonamido substances (PASFs):** This class includes polyfluorinated substances used as major raw materials. This class also includes PFAS which are environmental transformation intermediates and potential PFSA precursors. The carbon chains of these PFAS are perfluorinated (as suggested by the class name); however, they are considered polyfluorinated because the head group (defined below) of these PFAS contains carbon atoms bonded to hydrogen atoms. Thus, the alkyl carbons are not fully fluorinated. Representative compounds include N-ethyl perfluorooctane sulfonamidoethanol (N-EtFOSE) and N-ethyl perfluorooctane sulfonamido acetic acid (N-EtFOSAA).

- **Per- and polyfluoroalkyl ether carboxylates:** This class includes replacement chemistries for legacy C8 compounds, which are used as fluoropolymer processing aids. Two representative examples include ammonium 2,3,3,3-tetrafluoro-2-(heptafluoropropoxy) propanoate (HFPO-DA) and ammonium 4,8-dioxa-3H-perfluorononanoate (ADONA).

Differences in PFAS chemistries dictate how they partition between air, water, and soil/sediment phases as well as whether they are amenable to transformation processes. Fluorine characteristics and the strength of the carbon-fluorine bond contribute to PFAS being resistant to biodegradation, adsorption, oxidation, and other common treatment processes (ITRC 2022). The potential effects of these variable characteristics on PFAS media partitioning and behavior during treatment are summarized below.

- **Number of carbons:** Long-chain (PFCAs with eight or more carbon atoms or PFSAs with six or more carbon atoms) PFAS are easier to remove from the aqueous phase using sorption media technologies (i.e., GAC and AIX) than short-chain compounds (e.g., PFOA versus PFBA), due to their higher tendency to partition to solid surfaces and hydrophobic phases. Because long-chain PFAS tend to partition to solid phases, typically, they are preferentially found in biosolids relative to short-chain PFAS.

- **Proportion of fluorinated carbons:** The C-F bond is a strong covalent bond and difficult to cleave. Thus, many PFAS and associated terminal degradation products are thermally and chemically stable. As the proportion of fluorinated carbons increases, PFAS tend to be more difficult to completely degrade and more hydrophobic. PFAS with non-fluorinated carbon atoms (i.e., polyfluorinated, many of which may be referred to as precursors or intermediates) are susceptible to oxidative degradation, forming stable PFAA products.

- **Head (or functional) groups:** The head group (also known as the functional group) of PFAS can influence their physical and chemical properties. The head group also helps dictate which of the PFAS classes defined above apply. Examples of functional groups include carboxylates, sulfonates, phosphates, and amines. For treatment with GAC and AIX resin, sulfonates tend to be more readily absorbed than carboxylates with comparable chain-lengths.

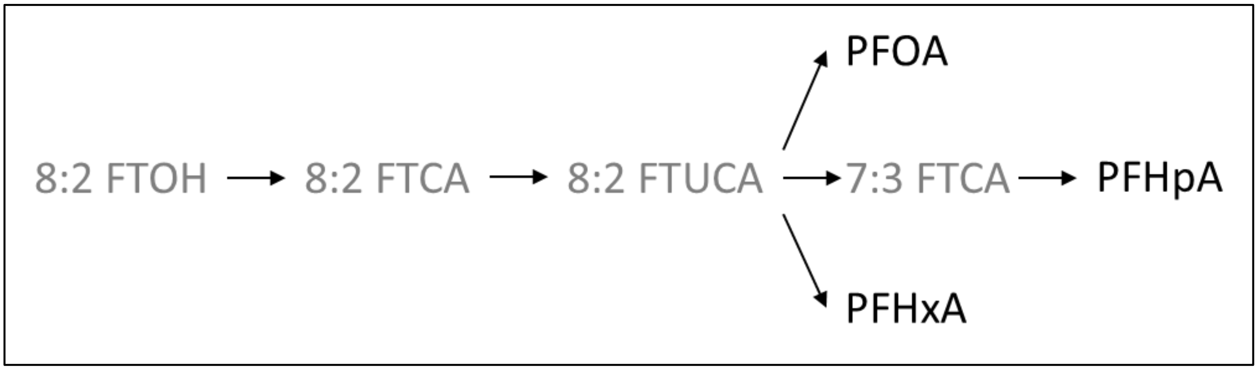
- **Acid versus base/anionic form:** Head groups of PFAS (e.g., carboxylates and sulfonates) can be protonated or deprotonated, which can impart either an uncharged form or a charged form (typically anionic) to the PFAS, respectively. The anionic form is more soluble in water and less likely to volatilize into the air phase than the uncharged, acid form. Based on typical acid dissociation constants (i.e., pK_a values) for PFAS and typical environmental pH conditions, most PFAS exist in the deprotonated, anionic form. Although, whether a PFAS is present in the environment in an uncharged or charged form depends on the chemistry of the specific PFAS and solution pH (ITRC 2021). For consistency throughout this report, the acid form will be used when referencing specific PFAS names (e.g., perfluorooctanoic acid rather than perfluorooctanoate).

2.2 PFAS Precursors and Transformations

PFAS precursors include several classes of PFAS, some of which were described in Section 2.1 (e.g., FASAs, fluorotelomer substances, PASFs, and polyfluoroalkyl ether acids). These PFAS typically contain non-fluorinated alkyl carbons that are amenable to microbial transformation processes, the products of which are typically terminal PFAAs.

Several WRRFs have reported an increase in total PFAS in the effluent compared to influent (Thompson et al. 2022; Houtz et al. 2016; Helmer, Reeves, and Cassidy 2022). Helmer et al. (2022) showed that effluent PFAS concentrations in 10 WRRFs were as much as 19 times greater than influent concentrations. Thus, WRRFs effectively transform polyfluorinated precursors into measurable, terminal degradation products, and long-chain PFAS preferentially adsorb to the biosolids during treatment (Helmer, Reeves, and Cassidy 2022).

The transformation of precursor or intermediate compounds to stable, measurable PFAS also occurs in land application of biosolids. The *U.S. EPA Land Application Field Study II* in October 2020 showed that on day one of land application, the PFAS in biosolids were dominated by intermediates and accounted for 95% of PFAS mass in the C8 path, as represented in Figure 2-1 (Herrmann et al. 2020). However, the ratio was reversed after 371 days of land application, where stable PFAS (i.e., PFOA, perfluoroheptanoate, perfluoroheptanoic acid (PFHpA), and perfluorohexanoate, perfluorohexanoic acid (PFHxA)) were dominant and accounted for 83% of the PFAS mass in the C8 path. The stable PFAS increased by 10 times for PFOA, 17 times for PFHpA, and 21 times for PFHxA over one year of land application. The transformation oxidative pathways are shown in Figure 2-1 (Herrmann et al. 2020).



8:2 FTOH is 8:2 fluorotelomer alcohol, 8:2 FTCA is 8:2 fluorotelomer carboxylic acid, 8:2 FTUCA is 8:2 fluorotelomer unsaturated carboxylic acid, 7:3 FTCA is 7:3 fluorotelomer carboxylic acid. (Herrmann et al. 2020).

Figure 2-1 Example PFAS precursor breakdown pathway to perfluoroalkyl end products

The TOP assay measures the total target PFAS and PFAS formation potential in wastewater and biosolids treatment processing. This assay involves oxidative treatment of a sample prior to PFAS measurement so that measured PFAS more closely reflect the end products of potential oxidative transformations. The influent TOP minus the sum of target PFAS in effluent and biosolids indicates residual target PFAS formation potential. In their Phase I Study, the California State Water Resources Control Board required PFAS sampling and analysis using the TOP assay for both influent and biosolids. The results presented to the Bay Area Clean Water Agencies Board meeting in August 2021 indicated the significant presence of PFAS precursors from the results of TOP (Lin, Mendez, and Sutton 2021).

Due to the transformation of precursors in wastewater influent to terminal PFAS in wastewater effluent and biosolids sources, the total oxidizable precursors (TOP) assay is strongly recommended to serve as a quantification method for precursor PFAS and suggested as a future “target” for wastewater and biosolids treatment in this study, along with selected measurable PFAS. However, studies reporting complete TOP results are limited. As a result, separation and destruction efficiencies for TOP precursors are not well defined, and TOP measurements were not included as a specific target in this study.

2.3 Selection of PFAS Considered for this Study

Due to the challenges of evaluating a large list of potentially relevant PFAS, target PFAS were selected for each waste stream to reflect a range of characteristics. This section presents the selected PFAS and the criteria and methodology used for their selection.

The following criteria were developed by MPCA and used in this Report to select the target PFAS:

- They are well described in the PFAS treatment literature.
- They have a diversity of functional groups (sulfonates and carboxylates).
- They have variable carbon chain lengths (four to eight).
- It is likely that treatment systems designed to treat these PFAS would also treat the majority of other PFAS that may also be present.
- Preliminary regulatory and/or guidance values have been established.

With these criteria in mind, Barr and Hazen reviewed the PFAS determined to be relevant to each waste stream at this time and either a) mapped them as being similar to an existing target PFAS or b) made it a new target PFAS. Table 2-1 lists target PFAS for each waste stream and typical concentrations assumed for this study. Appendix A outlines this process, along with chemical data for each of the 40 PFAS in U.S. EPA draft Method 1633 (US Environmental Protection Agency 2022). This list of commonly analyzed PFAS is a small fraction of the thousands of PFAS in use today, and has limited representation of replacement PFAS chemistries, many of which were not analyzed or not detected in available data sets.

The selected target PFAS include members from each class of PFAS presented in the ITRC guidance (ITRC 2021), with target PFAS for this study listed in bold:

- Perfluoroalkyl acids (PFAAs): **PFBA, PFBS, PFHxA, PFOA, and PFOS** reflect compounds with Minnesota Health Risk Limits (HRLs) or Health-Based Values (HBVs) and are included as target PFAS for all waste streams. These are sufficient to reflect the diversity in chemical characteristics in this class.
- Perfluoroalkane sulfonamides: **PFOSA** (perfluorooctanesulfonamide) was selected as a target PFAS for biosolids based on observed occurrence and concentrations.

- Fluorotelomer substances: **6:2 FTS** (6:2 fluorotelomer sulfonate) was selected because it is of moderate size, is in the MPCA 2020 guidance (MPCA 2020) and U.S. EPA Method 1633 (U.S. EPA 2021b) lists, and was relevant to multiple targeted waste streams.

- Perfluoroalkane sulfonamido substances: **N-MeFOSAA** (N-methyl perfluorooctanesulfonamido-acetic acid) was selected for biosolids because it is of moderate size and is in the MPCA 2020 guidance and U.S. EPA Method 1633 lists. **N-EtFOSAA** (N-ethyl perfluorooctane sulfonamidoacetic acid) was selected as a target PFAS for mixed MSW landfill leachate for similar reasons. In addition, more data are available for this parameter in leachate than N-MeFOSAA.

- Per- and polyfluoroalkyl ether carboxylic acids (replacement chemistries): No PFAS were selected from this class because, based on the current datasets available, no representative PFAS were detected in most samples reviewed for the four targeted waste streams despite these chemistries being increasingly used to replace other PFAS.

While replacement compounds are in development and active production to replace longer-chain PFAS, legacy compounds like PFOA and PFOS remain in widespread environmental circulation because no environmental mechanism exists to destroy them (Stoiber, Evans, and Naidenko 2020). Short-chain, replacement compounds have been less studied but have similar chemistry and potential human health impacts (Nian et al. 2020). As a result, target compounds include both legacy and replacement compounds.

Table 2-1 Summary of target PFAS and representative concentrations for this study

	Municipal WRRF Effluent[1]		Mixed MSW Landfill Leachate[2]		Compost Contact Water[3]		Municipal WRRF Biosolids[4]	
Basis	2009/2010 MPCA Data Helmer et al. (2022) Coggan et al. (2019)		Lang et al. (2017)		Wood (2019)		Venkatesan and Halden (2013) Helmer et al. (2022)	
Value Type	Typical (ng/L)	High (ng/L)	Typical (ng/L)	High (ng/L)	Typical (ng/L)	High (ng/L)	Typical (ng/g)	High (ng/g)
PFBA	**15**	**200**	**950**	**2,600**	**450**	**1,500**	**2**	**5**
PFBS	**15**	**50**	**250**	**650**	**10**	**25**	**20**	**40**
PFHxA	**30**	**60**	**1,500**	**4,000**	**500**	**2,000**	**5**	**15**
PFHxS	**5**	**15**	**350**	**750**	**10**	**100**	**15**	**30**
PFOA	**40**	**80**	**900**	**1,900**	**30**	**100**	**60**	**200**
PFOS	**5**	**30**	**150**	**300**	**20**	**1,500**	**400**	**600**
PFOSA	<2	<2	NA	NA	<10	<10	**20**	**70**
6:2 FTS	**5**	**150**	**150**	**350**	<20	<20	30	--[5]
N-EtFOSAA	ND	ND	**150**	**450**	<5	<5	**30**	**60**
N-MeFOSAA	ND	ND	NA	NA	<5	<5	**100**	**200**
Types of PFAS that are relevant	Short-chain PFCAs, long-chain legacy PFCAs and PFSAs, and fluorotelomers		Short and long-chain PFAAs, fluorotelomers, and precursors		PFCAs and PFSAs associated with composted items		Long-chain PFAAs that partition to biosolids, precursors	
Other characteristics of PFAS for each waste stream	Variable and reflective of consumer and industry changes		Variable and dependent on waste source, landfill age, and landfill type		High organic content and reduced; may have metals, other co-contaminants, and nutrients		High organic and nutrient content	

Bold and highlighted grey = target PFAS for each waste stream; ND=not detected, the detection or reporting limit was not specified in the referenced datasets; NA=not analyzed in source dataset; PFAA=perfluoroalkyl acids; PFCA=perfluoroalkyl carboxylates; PFSA=perfluoroalkane sulfonates

[1] Data reflect aqueous concentrations in treated effluent from municipal WRRFs.

[2] Data reflect aqueous concentrations in leachate from mixed MSW landfills.

[3] Data reflect aqueous concentrations in surface water that have come in contact with waste materials at commercial composting facilities.

[4] Data reflect concentrations in biosolids material as it leaves the municipal WRRF before any off-site processing or storage.

[5] Based on one data point from one study, so no "high" value developed.

2.4 Treatment Target Selection

For this Report, MPCA selected "non-detectable" concentrations of target PFAS as a treatment goal to guide the alternative evaluation. This stance of targeting analytical reporting limits is aggressive, especially for short-chain PFAS with limited regulatory guidance; many beneficial projects may instead target mass removal or percent removal of total PFAS or long-chain PFAS instead of a specific concentration target.

However, evolving regulations around PFAS risk-based limits make it challenging to select alternative treatment targets.

Targeting “non-detectable” effluent concentrations may also be moving target over time, as analytical methods with the ability to measure lower and lower concentrations are developed. For this study, treatment targets were set by MPCA as current analytical reporting limits. MPCA’s document “Guidance for Per- and Polyfluoroalkyl Substances: Analytical” is intended for use starting in 2023 (MPCA 2020) and includes reporting limit goals of approximately 5 ng/L for aqueous and 5 ng/g for solid matrices. Current analytical reporting limits for both aqueous and solids materials are in that range. **This study uses 5 ng/L and 5 ng/g as treatment targets.** While some labs are achieving lower reporting limits, differences within approximately one lower order of magnitude would not necessarily affect technology selection for this study.

3 Selection of Currently Feasible Treatment Technologies for PFAS Removal

This section describes the individual technologies considered for this study and the screening approach used to identify currently feasible treatment technologies for PFAS removal. Currently feasible treatment technologies are defined here as technologies that have been implemented at scales similar to those considered in this study and demonstrated to be effective for PFAS removal and/or destruction (i.e., there is a publicly available body of work demonstrating technology effectiveness).

3.1 Technology Screening Approach

A comprehensive list of potential treatment technologies for PFAS, considerations for their full-scale implementation, and their technology readiness level are in Appendix B. Each technology was screened against two threshold criteria, as described below.

- Demonstrated treatment scale: Technologies passing this screening criterion must have been “field-implemented” and commercially available for liquid technologies or implemented at the pilot-scale for biosolids. Two different standards were used here for liquid streams and biosolids. This is because established technologies have been widely applied at full scale for liquid-phase PFAS treatment, while biosolids have only been treated at the demonstration or pilot scale.
- Demonstrated PFAS removal: Technologies passing this screening criterion must have publicly available data demonstrating at least 90% separation or destruction of at least one selected PFAS (or removal to below method-reporting limits for that study).

The ability of each technology to meet the two screening criteria is described and rated in Appendix B. The appendix table includes a “yes” or “no” assessment of whether a technology meets each criterion. Technologies that were assigned a “yes” for both screening criteria are described in more detail in the following subsections. These technologies were carried forward for assembly into a PFAS management alternative, as described in Section 4, and further evaluated for inclusion in preliminary design and cost-estimating efforts.

The field of PFAS separation and destruction technologies is rapidly evolving. The results of this technology screening exercise reflect publicly available information in December 2022. Barr and Hazen acknowledge that some of the technologies included on the initial compiled list have demonstrated PFAS removal but are not yet commercially available or may become available in the near future. Despite being promising, these technologies are outside the scope of this study and, thus, were not carried forward. Section 11.7 further discusses technology readiness and the future of PFAS management technologies.

3.2 Technology Screening Results—PFAS Liquid-Liquid Separation

The following liquid-liquid PFAS separation technologies passed the threshold screening process and were considered currently feasible PFAS management alternatives. These technologies produce two separate liquid streams: a treated water stream (large volume with lower concentrations of PFAS than the feed water) and a waste stream (small volume with higher concentrations of PFAS than the feed water). These technologies are commonly used to concentrate PFAS into a smaller volume of water prior to further treatment with other PFAS separation or destruction technologies. These technologies include nanofiltration/RO (NF/RO) membrane separation and foam fractionation.

3.2.1 NF/RO Membrane Separation

NF/RO membrane separation concentrates PFAS in the feed stream into a lower volume concentrate stream by physical separation via high-pressure membranes. NF membranes typically have higher water recovery or operate at lower pressures than RO due to larger membrane pore sizes. RO membranes have proven more effective at PFAS separation because of their more dense construction and smaller effective pore size (Jin, Peydayesh, and Mezzenga 2021). Operation of an NF/RO membrane system typically requires, at minimum, pretreatment with micro- or ultrafiltration (MF and UF, respectively) to protect the NF/RO membranes from suspended solids fouling. Additional pretreatment may be required depending on the specific water quality conditions of the waste stream. MF and UF water recoveries depend on the feed water quality and the frequency of backwashes and chemical cleanings but are typically on the order of 80–98%. NF/RO recovery depends on the influent concentrations of potentially scale-forming parameters, operating conditions, and membrane construction. Recoveries range from 50–80% for conventional RO to 90% for high-recovery RO technologies (Crittenden et al. 2012). The waste stream generated by membrane systems for PFAS treatment can be as large as 5–30% of the treated flow rate (AWWA 2019; NGWA 2017).

Specialty RO membrane systems also exist that may limit the pretreatment required for liquid waste streams with high fouling potential (due to organics and/or mineral concentrations) compared to conventional RO. These systems are typically constructed with wider feed spacers than conventional RO, which improves the flow distribution of the feed, limits the pressure drop across the membrane in streams with high fouling potential, and allows suspended solids to flow between membrane layers more easily (Bates, Bartels, and Franks 2008). The wide spacers also allow for more effective chemical cleaning of the membranes to remove foulants during routine clean-in-place procedures. Specialty RO membrane systems are most applicable for sites with low-to-moderate flow (e.g., landfill leachate). They are not expected to be cost-competitive with conventional RO for high-flow systems like WRRF effluent. Common approaches for management of wide-spaced RO concentrate are (1) solidification and solids disposal in the landfill, (2) reuse as a wetting and compaction agent in the landfill, or (3) deep well injection, none of which destroy PFAS as targeted in this study.

PFAS removal or “rejection” across membranes depends on influent pH (Steinle-Darling and Reinhard 2008), transmembrane pressure, the characteristics of other ions in solution (Zhao et al. 2013), and the concentration, size, and hydrophobicity of each specific PFAS compound (Wang et al. 2018). Higher operating pressures and associated recovery typically correlate to higher rejection rates for PFAS because they result in increased water flux with limited change in PFAS transfer (Jin, Peydayesh, and Mezzenga 2021; Bellona et al. 2004; Tang et al. 2007). Demonstrated PFOS separation (membrane rejection) is over 99% for RO membranes and 90–99% for NF membranes (ITRC 2022). One study found over 93% rejection of PFAA as small as PFBA using a cross-flow NF membrane (Appleman et al. 2013), but other studies report much lower rejection rates for short-chain PFOS, down to 20% for PFBS (Wang et al. 2018). While data are more limited, specialty wide-spaced RO systems have also demonstrated high rejection of PFAS in landfill leachate. PFOS, PFOA, PFHxS, PFHxA, PFBS, and PFBA have all shown >99% rejection based on the data available (Stanford 2019). Appendix C compares NF and RO rejection for different PFAS across multiple studies.

The removal of dissolved minerals by RO systems may increase the corrosivity of the treated water (low pH, hardness, and alkalinity), and post-treatment mineral addition may be required (ITRC 2018; NGWA 2017). NF has lower rejection rates for monovalent anions, like chloride or sodium, and thus may require less post-treatment than RO (ITRC 2018). Thermal evaporation and crystallization are thermal processes that evaporate water, leaving a concentrated solid residual and reducing the concentrated RO/NF brine volume. Thermal evaporation and crystallization were not considered in this study for brine management due to the potential for PFAS partitioning from the aqueous brine to the vapor or gaseous stream. If PFAS do partition to the vapor/gaseous stream, removing PFAS from that stream could require a thermal oxidizer. Ultimately, the combination of evaporation/crystallization with thermal oxidation of the gas stream resembles direct incineration of the brine, assuming the solid residual would be managed by incineration.

3.2.2 Foam Fractionation

Foam fractionation is a separation technology that concentrates PFAS from a water phase into a smaller volume foam phase using fine air bubbles. This technology takes advantage of the surfactant properties of PFAS, enabling PFAS to partition to the surface of rising air bubbles (Smith et al. 2022). The resulting foam is collected and collapsed (referred to as the foamate) for further treatment or management off-site or, in the future, could be managed by an on-site destruction technique (Buckley et al. 2022). Foam fractionation can be an advantageous treatment alternative for PFAS because treatment systems are relatively simple and have relatively low capital costs, energy requirements, and operational costs (Buckley et al. 2022). In addition, produced foamate volumes are significantly lower than RO/NF concentrate volumes with concentration factors up to one million (Burns, Stevenson, and Murphy 2021), much higher than RO/NF concentration of approximately two times (50% recovery) to 10 times (90% recovery). Conventional aeration basins could potentially be retrofitted as a foam fractionation system (Smith et al. 2022).

Commercial foam fractionation systems are typically operated in continuous batches (i.e., batches are sequenced and run in parallel). PFAS removal depends on several operational parameters such as contact time, air ratio, air flow rate, and foam fraction (Dai et al. 2019; Smith et al. 2022). Removal efficiencies for long-chain PFAS are typically greater than 90% (Meng et al. 2018; Robey et al. 2020; Burns et al. 2022). Removal of short-chain PFAS is typically 50% or less. For example, PFBA and PFBS tend to have removals of less than 10% (Smith et al. 2022). The use of foaming agents, such as cationic surfactants, can improve short-chain PFAS removal, but overall removal is still relatively low (Beattie, Salvetti, and Macbeth 2022; Vo et al. 2023). Vacuum pumps on the foam collection system may also improve short-chain removal. Smith et al. (2022) also observed that mass balances for long-chain PFAS did not close through the foam fractionation system, potentially suggesting that mass is lost as aerosol. Foam fractionation removal efficiency for different PFAS is compared across multiple studies in Appendix C.

Ozone may be used instead of air and may improve separation efficiencies due to a relatively higher affinity of the PFAS head group to ozone than air. However, this chemistry is somewhat uncertain (Buckley et al. 2022). Ozone may also degrade PFAS precursors into terminal PFCAs. Dai et al. (2019) demonstrated up to 80% PFAS removal without ozone and up to 95% PFAS removal when combined with ozone, with both having relatively low removal for short-chain PFAS.

Based on vendor-provided information, current full-scale foam fractionation installations can treat flows of 26,000–132,000 gallons per day (gpd) or approximately 20–100 gallons per minute (gpm), including one 60 gpm field trial (Burns et al. 2022).

3.2.3 Comparison of PFAS Liquid Separation Technologies

NF/RO membrane separation is a well-developed technology deployed across multiple industries and able to treat water flow rates from several gallons per minute to several MGD. While foam fractionation is an effective technology for PFAS, the size of demonstrated applications is lower, on the order of 20–100 gpm. It is not currently feasible as a primary separation technology for large municipal WRRFs, but it may be feasible for small WRRFs, mixed MSW landfill leachate, and compost contact water installations. Foam fractionation is a more specific treatment option for PFAS than NF/RO. It has the potential for lower operational costs, fewer pretreatment requirements, and higher recovery of treated water than NF/RO but is not very effective at separating short-chain PFAS. In contrast, RO/NF can reject multiple potential contaminants of concern providing benefits for potential future contaminants of interest.

3.3 Technology Screening Results—PFAS Liquid-Solid Separation

The following liquid-solid PFAS separation technologies passed the threshold screening process and were considered currently feasible PFAS management alternatives. These technologies transfer PFAS from liquid to solid media producing a treated water stream with low concentrations of PFAS and a solid media phase requiring further treatment (destruction, regeneration, or reactivation) or disposal.

3.3.1 Granular Activated Carbon Adsorption

GAC adsorption is a separation technology that removes PFAS from the liquid stream through adsorption to the solid surface of the GAC media. GAC is one of the most common treatment technologies for removing PFAS from water (Darlington, Barth, and McKernan 2018), can be rapidly deployed, and typically requires modest operator involvement (NGWA 2017). Virgin bituminous coal-based GACs are typically used for PFAS removal due to their higher removal performance. However, enhanced virgin coconut-based GACs may be applicable in some installations as well as reactivated GAC (see Section 3.3.2).

PFAS removal occurs via a physical mass transfer process (ITRC 2018) between the non-polar functional groups on the GAC and the hydrophobic tail of the PFAS (Darlington, Barth, and McKernan 2018). Due to the adsorption mechanism, GAC can remove PFAS to below detection limits until breakthrough occurs.

PFAS removal efficiencies can be higher than 99%, with effluent concentrations near or below analytical detection limits (AWWA 2019).

However, GAC does not adsorb all PFAS equally. GAC has higher removal capacities for long-chain PFAS and sulfonates than short-chain PFAS and carboxylates (ITRC 2018). Because the GAC media has a finite number of adsorption sites, readily sorbed, long-chain PFAS can displace the short-chain PFAS on the GAC surface, leading to faster breakthrough times for short-chain carboxylates, such as PFBA.

While short-chain PFAS have lower GAC loading capacities and faster breakthrough times than long-chain PFAS, they can be effectively removed if the GAC media is replaced at higher frequencies. The required GAC changeout frequency and associated operational cost depend on the effluent targets, influent PFAS concentrations, influent concentrations of other compounds adsorbing to GAC, and the vessel configuration. The effluent PFAS concentration is dependent on the empty bed contact time (EBCT) of the GAC reactor (commonly between 10 and 20 minutes (AWWA 2019)), how frequently the GAC is replaced, influent PFAS concentrations, and the general feed water quality. The difference in duration between the time to breakthrough of PFBA and a long-chain PFAS like PFOA varies by water quality and influent concentrations but has been observed to be between two and 10 times more rapid for four-carbon PFAS than for the most similar eight-carbon PFAS (Westreich et al. 2018; J. Burkhardt et al. 2019; Franke et al. 2021). This suggests that the frequency of GAC changeout when targeting PFBA rather than PFOA would need to occur much more frequently. Bed volumes to the first breakthrough are compared for different PFAS across multiple studies in Appendix D.

The efficacy of PFAS removal by GAC can be inhibited by competitive adsorption (NGWA 2017). High concentrations of total organic carbon (TOC) and other hydrophobic compounds (such as volatile organic compounds) in the water can compete for GAC adsorption sites, which reduces PFAS adsorption capacity and may increase changeout frequency (AWWA 2019; Darlington, Barth, and McKernan 2018). Bench-scale accelerated column testing or pilot testing using site-specific water is recommended to estimate the expected GAC changeout frequency. High concentrations of iron, manganese, or particulates can also foul GAC media and cause short-circuiting in the GAC bed (AWWA 2019) or pressure increases that trigger GAC replacement. Bacterial growth can also occur in the presence of sufficient nutrients and cause pressure increases that trigger backwashing, disinfection, or GAC replacement.

GAC systems for PFAS are commonly operated as pressure vessels in series, using a lead-lag configuration that enables improved treatment efficiency and PFAS breakthrough monitoring PFAS between the lead and lag vessels. Vessels are operated in a downflow configuration under pressure, with adsorption occurring at the top of the bed and gradually moving down the bed as the adsorption sites on the GAC are filled. The area of the GAC bed where active adsorption occurs is known as the mass transfer zone. The depth of the mass transfer zone and the rate at which the mass transfer zone travels through the GAC bed differs for different PFAS. Typically, the concentrations of PFAS are monitored between the lead and lag vessels, and when the mass transfer zones of PFAS reach the bottom of the lead bed, PFAS are detected in the effluent of the lead vessel. Once a treatment threshold has been reached, the GAC media in the lead vessel is replaced, and the locations of the lead and lag vessels are switched such that the former lag vessel becomes the new lead, and the former lead vessel becomes the new lag. Monitoring frequency depends on estimated changeout frequency and PFAS treatment targets.

Current options for spent media disposal include thermal reactivation, incineration, or landfilling. Thermal reactivation and incineration are discussed in more detail in Section 3.4.

3.3.2 Reactivated Granular Activated Carbon

Reactivated GAC is GAC that has undergone a thermal reactivation process to restore its sorption capacity after being used. The GAC reactivation process and PFAS destruction efficiency are discussed further in Section 3.4.2.

Thermal GAC *reactivation* occurs at 980°C and differs from thermal GAC *regeneration,* which is conducted at 200°C using either steam or nitrogen gas (DiStefano et al. 2022). GAC *regeneration* can also be completed using a solvent/brine (Siriwardena et al. 2021). These GAC regeneration options remove sorbents from the GAC surfaces, but for PFAS destruction, additional treatment would be needed.

Thermally reactivated GAC can generally be categorized into two types:

1. General pool reactivated GAC: This is GAC that is pooled from multiple sources after use, thermally reactivated, and resold for reuse. The grade and quality of this material can be variable. It is not typically used for PFAS treatment, although it can be used as part of the treatment train to sorb background organics.

2. Site-dedicated reactivated GAC: This is GAC that is segregated by site during the reactivation process. By segregating the GAC, the grade and quality of the GAC can be better controlled. Time to PFAS breakthrough is similar between virgin GAC and site-dedicated, reactivated GAC but lower for general pool, reactivated GAC (McNamara et al. 2018; Westreich et al. 2018). Due to the size of commercial reactivation facilities, site-dedicated reactivation is limited to facilities that produce approximately 80,000 pounds of GAC or more per media changeout event.

The benefit of thermal reactivation of GAC is that the GAC media can be reused, reducing ongoing operational costs associated with a GAC treatment system. Based on vendor-provided information, after the initial purchase of GAC media, ongoing GAC replacement costs can be up to 50% lower for reactivation and reuse of GAC compared to costs to dispose of spent media and purchase new media for single-use GAC. For this study, site-dedicated, reactivated GAC is considered for facilities expected to generate 80,000 pounds of GAC or more per media changeout event. General pool reactivated GAC is not considered a PFAS separation technology due to the expected rapid PFAS breakthrough.

3.3.3 Single-Use Anion Exchange Resin

Single-use AIX resins for PFAS removal are typically positively charged polystyrene beads with tertiary amine, quaternary amine, or complex amino functional groups (Boyer, Fang, et al. 2021). AIX resins remove PFAS from the liquid stream via two primary mechanisms:

- Electrostatic interactions between the negatively charged head groups of PFAS and AIX resin cationic functional groups, and
- Hydrophobic interactions between the fluorinated alkyl chain of PFAS and the AIX resin structural polymer backbone (Boyer, Fang, et al. 2021).

Single-use ion exchange refers to using AIX resin once, then replacing the AIX resin and disposing of it via landfill or high-temperature incineration.

Like GAC, AIX resin is typically applied in lead-lag pressure vessels. AIX can remove many PFAS to below detection limits until breakthrough occurs. Also similar to GAC, long-chain PFAS within each PFAS subgroup (ITRC 2022) are more readily removed and thus have a longer treatment time or volume of water treated before breakthrough occurs. PFAS removal efficiencies with AIX can be upwards of 99% with new resin (AWWA 2019) but change as the resin is used and the PFAS sorptive and exchange capacity is exhausted. Boyer et al. (2021) summarized bed volumes treated to 10% breakthrough as reported in the literature for various water matrices and individual PFAS. Time to breakthrough depended on several variables, including the type of AIX resin used, EBCT, initial PFAS concentrations, concentrations of major anions (such as sulfate), and concentration of natural organic matter. The kinetics of PFAS uptake by AIX resins is typically faster than GAC (ITRC 2022), which results in shorter design EBCTs on the order of two to five minutes (AWWA 2019). Generally, the breakthrough of PFSAs with AIX resins occurs after 10,000–100,000 bed volumes or more, whereas the breakthrough of PFCAs with AIX resins occurs after 1,000–100,000 bed volumes (Boyer, Fang, et al. 2021). Bed volumes to first breakthrough are compared for different PFAS across multiple studies in Appendix D.

Anions can inhibit AIX media effectiveness; for example, nitrate, sulfate, and perchlorate can compete with PFAS for the AIX sites and should be removed from the water before PFAS treatment using AIX (NGWA 2017). In addition, iron and manganese can foul AIX resin, similar to GAC, and cause short-circuiting (AWWA 2019). Organics and minerals can also block pore spaces in the resin. Pretreatment is required for waters with these co-contaminants. In particular, removing negatively charged ions can be expensive, often requiring targeted ion exchange media or membrane treatment as a pretreatment.

3.3.4 Regenerable Anion Exchange Resin and Solvent Regeneration

Regenerable AIX operates the same way as single-use AIX, but once exhausted, the AIX resin is regenerated using a brine/solvent mixture and returned to service. The regeneration efficiency depends on the specific resin used and the regeneration solution used (Boyer, Fang, et al. 2021). Regeneration efficiency can be as high as 90% or more for PFOS and PFOA (Boyer, Fang, et al. 2021). The removal capacity of regenerable AIX resin is generally less than single-use AIX resin (ITRC 2022), requiring regeneration more frequently than single-use AIX changeout. The lifespan and long-term performance of regenerable resin for PFAS management is uncertain because the first publicly known regenerable AIX system was installed in 2018 (ITRC 2022). If the resin is not rinsed properly before being put back into service, it can cause corrosion issues downstream (ITRC 2018).

The benefits of being able to regenerate the resin can offset the costs of purchasing single-use resin, especially for larger systems. Based on vendor-provided information, typical costs to regenerate AIX resin are $5–$100 per cubic foot of resin (compared to $300–$400 per cubic foot for new resin purchase). Regeneration of AIX resin will save on media costs but requires larger vessels than single-use AIX (although still smaller than GAC vessels for comparable flows). The typical EBCT for regenerable AIX is 5–6 minutes, slightly longer than single-use resin, because regenerable AIX resins have approximately one-third the sorptive capacity of single-use resins. Smaller sites using off-site regeneration will have slightly higher transportation costs for the higher media volume required compared to disposal of single-use resin (assuming similar travel distances). Some facilities may be large enough and have enough media to warrant the additional cost of constructing an on-site AIX resin regeneration facility to reduce long-term operating costs. The cost-benefit of off-site versus on-site media regeneration is site-specific.

Regeneration can be completed on-site or in a regional facility. AIX regeneration involves a solvent brine solution that helps desorb PFAS from the resin. This brine can be thermally distilled, with PFAS routed to a small volume of still bottoms or loaded onto a small volume of GAC at very long EBCT. Resulting still bottoms or GAC can be subjected to destruction using any of the technologies outlined in Section 3.4.

3.3.5 Modified Clay Adsorption

Modified clays have a high surface area and small uniform pore size (ITRC 2022). They are typically produced from smectite clays treated with quaternary ammonium surfactants (Yan et al. 2020; Jiang et al. 2022). They remove PFAS using the dual mechanism of adsorption and ion exchange, similar to AIX media. Modified clays can be naturally occurring or made with surface modifications that incorporate electrostatic and hydrophobic moieties specific to PFAS treatment (ITRC 2022). Modified clay media is used in a fixed bed pressure vessel and can be operated in lead-lag arrangement, similar to GAC and AIX operations. Once exhausted, the clay media is removed and disposed of through landfilling or high-temperature incineration.

Modified clay media have been used to demonstrate removal of PFAS to non-detect concentrations, but adsorption performance varies by clay media type and surface treatments (Pannu and Plumlee 2021). Modified clay adsorption treatment has been used to treat groundwater AFFF contamination in pump-and-treat systems. A study using modified clays to remove AFFF from groundwater found PFAS removal efficiencies of 95–99% (Yan et al. 2020). Modified clay absorbents have also been used for landfill leachate, groundwater, stormwater, and drinking water treatment (Goldenberg 2022). Modified clay media have demonstrated longer bed life than GAC for short-chain PFAS, which are typically enriched in landfill leachate (Donovan 2022). In the Orange County Water District pilot test, a modified clay adsorbent had the longest life of any of the sorption media tested, which also included GAC and AIX (Pannu and Plumlee 2021).

Salts can precipitate on the clay media and effectively blind it from the PFAS. Based on vendor-provided information, modified clay media can foul with concentrations of salts above 5,000 mg/L TDS but will continue to remove PFAS until concentrations reach 16,000 mg/L TDS. Laboratory studies have shown high PFAS removals with TOC concentrations of up to 1,300 mg/L (Goldenberg 2022), suggesting PFAS removal by modified clay is less subject to competition with background organic material than removal by

AIX or GAC. However, modified clays are still a filtration media in a pressure vessel. They are expected to have similar pretreatment requirements for solids and metal foulants to prevent increased operating pressure due to pore space blockage.

3.3.6 Comparison of PFAS Phase Separation Technologies for Liquid Streams

Each of the five sorption media passing the technology screening effectively removes PFAS from liquid waste streams. Sorption media should be selected based on influent water quality, pretreatment requirement, short-chain PFAS removal needs, treatment goals, and cost. GAC and AIX are both widely used for PFAS removal, and modified clay has been applied to a lesser degree. GAC is often used instead of AIX because less pretreatment is required. With the information available, modified clay media requires even less pretreatment than GAC. GAC is not very effective for short-chain carboxylic acid PFAS such as PFBA and requires frequent changeout to remove them consistently. AIX is sometimes selected due to its higher sorption capacity, especially for shorter PFAS, so AIX systems are expected to have a lower media use rate than GAC systems. The limited data for modified clay applications suggests that their sorption capacity is more similar to AIX than GAC. Generally, on a per unit volume basis, AIX resin is more expensive than modified clay, and modified clay is more expensive than GAC media.

Regenerable AIX is typically selected over single-use AIX when PFAS concentrations and predicted changeout frequencies are high. Regenerable AIX has a lower sorption capacity than single-pass AIX but can be regenerated on-site at a specialized regeneration facility and reused, which reduces the cost of operation compared to single use AIX resins (Bolea 2022).

Reactivated GAC is typically selected over GAC when a large volume of GAC is required, making it possible to maintain dedicated, site-specific regeneration. Using current vendors, a site would need approximately 80,000 pounds of GAC in circulation for dedicated reactivation, which correlates to a WRRF over 1 MGD in size based on conceptual equipment sizing and estimated changeout frequency.

3.4 Technology Screening Results—PFAS Destruction

The technologies described in this section are intended to mineralize PFAS to non-fluorinated products, requiring the breakage of carbon-fluorine bonds. These technologies rely on high temperature or a combination of high temperature and pressure to mineralize PFAS. They include thermal reactivation of GAC, high-temperature incineration (also known as hazardous waste incineration), SCWO, pyrolysis with thermal oxidation, and gasification with thermal oxidation.

The first two listed, as well as thermal oxidation, were detailed as potential thermal treatment options in the U.S. EPA's *Interim Guidance on Destroying and Disposing of Certain PFAS and PFAS-Containing Materials* (U.S. EPA 2020). SCWO, pyrolysis, and gasification are less developed but were in this study as the most developed biosolids management options capable of achieving near-complete PFAS destruction.

Regulatory and scientific information on the efficacy of PFAS destruction in high-temperature incinerators and other thermal, non-hazardous waste combustors remains uncertain. Areas of uncertainty include the potential formation of products of incomplete combustion (PICs); unknown occurrence of PFAS or PICs in emission control devices; lack of standardized methods to monitor potential gaseous PFAS emissions, especially for less common PFAS; and variability in destruction efficiency due to different incinerator designs, waste feed approaches, and operational conditions (temperatures, residence times, and turbulence) (U.S. EPA 2020).

The U.S. EPA recently evaluated the efficacy and technology readiness of non-combustion-based technologies potentially capable of mineralizing PFAS associated with spent GAC and AIX media, soils, biosolids, AFFF, and landfill leachate (Berg et al. 2022). This study identified limited technological readiness outside of high-temperature incineration, except potentially for applying SCWO for AFFF destruction and pyrolysis for biosolids treatment (Berg et al. 2022). Limited data are currently available to assess the applicability of PFAS destruction technologies for the waste streams considered, largely due to the areas of uncertainty identified above.

Furthermore, on April 26, 2022, the U.S. Department of Defense (U.S. DOD) issued a temporary prohibition on the incineration of U.S. DOD PFAS waste streams, including AFFF and water treatment media. Some states, including Illinois and New York, have also placed a moratorium on the incineration of PFAS-containing waste streams.

3.4.1 High-Temperature Incineration (retained for disposal of Sorption Media and Concentrates)

Incineration is combustion in the presence of excess air. Two common incinerators are sewage sludge incinerators (SSIs) and hazardous waste incinerators (HWIs). To achieve thermal decomposition and destruction of PFAS exceeding 99.9%, temperatures exceeding at least approximately 500–700°C are required (Xiao et al. 2020). These temperatures are above the typical operational range of SSIs (L. J. Winchell et al. 2021a). PFAS incineration at lower temperatures might be possible with longer residence time, mixing, and favorable stoichiometry, but much work is still needed to understand the conditions for complete combustion (Lewis 2008; Niessen 2002; L. J. Winchell et al. 2021a).

In contrast to SSIs, HWIs are typically operated at higher temperatures (650–1,650°C) and longer solids residence times (0.5–1.5 hours) (U.S. EPA 2020). Currently, approximately 10 HWI facilities are operating in the United States (U.S. EPA 2020). HWIs typically accept solid or liquid wastes and consist of a rotary kiln with an afterburner for further gas phase thermal destruction, along with ancillary equipment for air pollution control, influent gas conditioning, ash treatment, scrubber blowdown, management, and boilers. Afterburners are typically operated at 1,100–1,370°C with a gas phase residence time of one to three seconds (U.S. EPA 2020). Afterburners at this temperature can control PFAS at an efficiency greater than 99% (Barranco, Caprio, and Hay 2020).

Commercial incineration of sorption media, such as GAC and AIX, is a common practice, though limited research has been published on its performance. Available research suggests efficient thermal decomposition of many target PFAS sorbed to GAC (Xiao et al. 2020). Additional research is needed to verify PFAS destruction and evaluate the formation of PICs, including other PFAS not commonly measured (U.S. EPA 2020).

3.4.2 Thermal Reactivation of Granular Activated Carbon (retained for disposal of Sorption Media)

PFAS can be desorbed from GAC and destroyed using thermal GAC reactivation. The process restores most of the sorptive capacity of the media. The process of thermal GAC reactivation includes:

- Exposing GAC to temperatures up to 980°C in a multiple hearth furnace or rotary kiln furnace under low oxygen conditions. PFAS and other sorbents are volatilized and partially destroyed in the furnace.
- Volatilized PFAS and other sorbents not completely combusted in the furnace are drawn into an abatement system consisting of a thermal oxidizer (afterburner), a scrubber, and a baghouse.

PFAS separation during GAC reactivation is PFAS- and temperature-specific. Decomposition of 80–100% of PFCAs and PFSAs occurs at 500°C, and up to 80–100% defluorination of both PFCAs and PFSAs has been observed at 1,000°C (Sonmez Baghirzade et al. 2021). One test evaluating PFAS sorbed to GAC before and after thermal reactivation found that of 21 PFAS detected on the spent media, all PFAS, except for PFDA, were below analytical detection limits (0.5–2.9 ng PFAS/g GAC) after reactivation (DiStefano et al. 2022). In the stack emissions, 23 of 36 target PFAS were detected. On a mass-basis, a 99.99% removal efficiency was observed between the mass of PFAS entering the reactivation furnace and the mass of PFAS leaving through the stack emissions (DiStefano et al. 2022). The performance of reactivated GAC for removal of PFAS via adsorption was previously described in Section 3.3.2.

3.4.3 Supercritical Water Oxidation (retained for disposal of Biosolids)

Any material containing water subjected simultaneously to high temperature and pressure (374°C and pressure of 250 bar, respectively) will reach the supercritical state for water, where most organic material dissolves. Organic material is then oxidized through the addition of air, resulting in byproducts of CO_2, minerals, water, and energy (if the feedstock supports it). This process can be economical for the disposal of biosolids, sorption media, or any organic waste (e.g., AFFF), but fuel costs make direct SCWO of dilute liquid waste streams impractical. SCWO has been successfully implemented commercially for hazardous materials and is currently undergoing demonstration pilots for PFAS-specific applications (Krause et al. 2022). Figure 3-1 shows an example process flow diagram of an SCWO system.

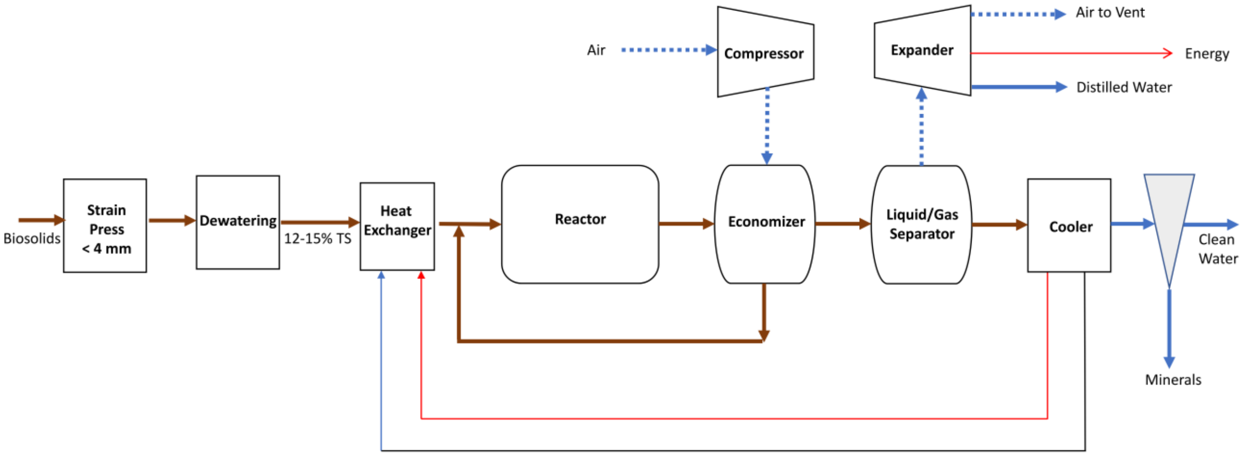


Figure 3-1 Example SCWO process flow diagram

PFAS removal and transformation in SCWO reactors depend on the design and operational parameters such as batch or continuous flow, excess oxygen, and reaction time. Research to date has established the successful destruction of PFAS using SCWO. For example, in one study, the destruction of up to 70% PFOS was observed in a batch reactor (Pinkard et al. 2021), but the same group saw 99.999% PFOS destruction in a continuous reactor (J. Li et al. 2023). In another study, 99% or greater destruction of PFOS and PFOA was observed using diluted AFFF samples (initial concentrations on the order of mg/L to final concentrations on the order of µg/L) (Krause et al. 2022).

Based on vendor-provided feedback, SCWO can be operated as a self-sustaining reaction with a minimum influent energy content of 2–3 MJ/kg (1,000 BTU/lb) and can produce electricity with some feedstocks, including biosolids and possibly GAC or AIX media slurries. This minimum self-sustaining energy requirement translates to approximately 100,000 mg/L of chemical oxygen demand (COD) in water phase wastes. Feedstocks with lower energy content would require a co-fuel, such as diesel or isopropanol, which quickly makes the treatment of many water streams uneconomical.

3.4.4 Pyrolysis with Thermal Oxidation (retained for disposal of Biosolids)

Pyrolysis is the thermo-chemical decomposition of organic matter at high temperatures without oxygen. Pyrolysis of biosolids is achieved by heating the material to 500–800°C in a reactor purged with nitrogen gas to remove oxygen. The biosolids are the fuel for the thermal decomposition reaction and must be dewatered to contain less than 25% moisture before pyrolysis. During pyrolysis, the solid molecules vibrate rapidly and break apart into lighter volatile molecules and fixed carbon, producing pyrogas, tars, oils, and biochar (Paz-Ferreiro et al. 2018). Figure 3-2 shows an example flow diagram for a pyrolysis unit.

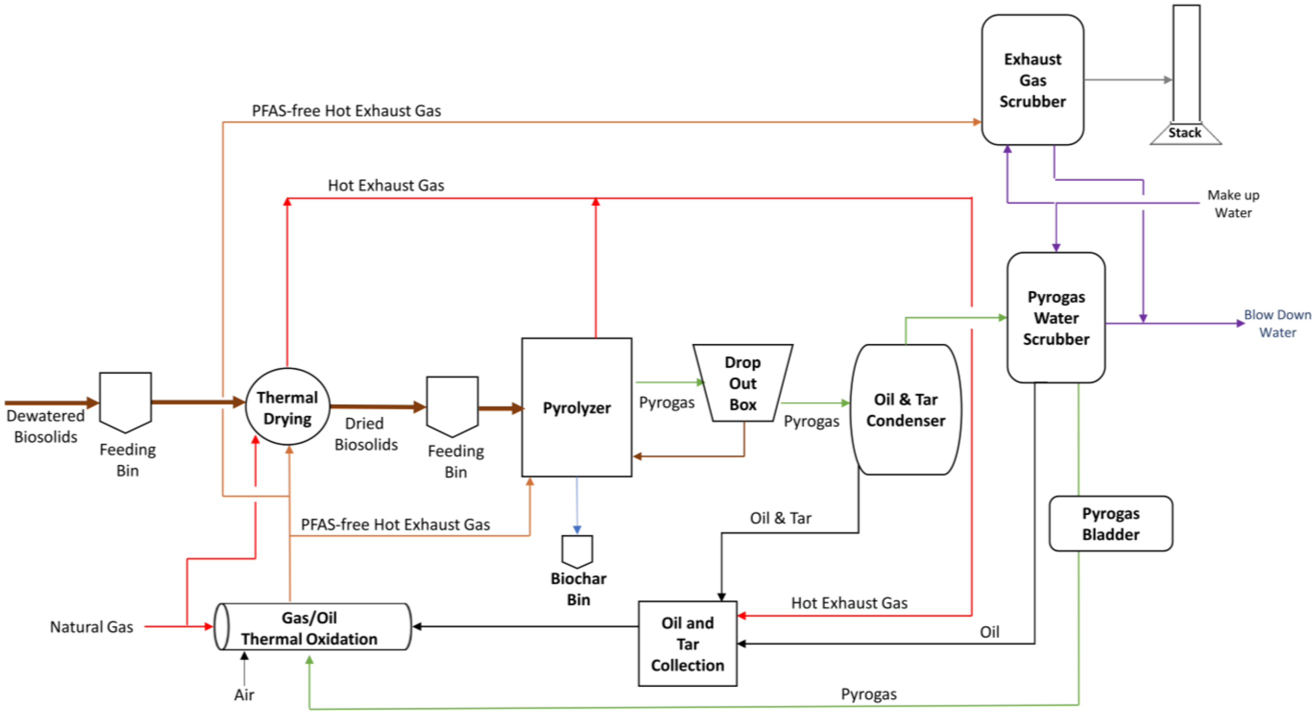


Figure 3-2 **Example pyrolysis process flow diagram**

The exhaust gas from pyrolysis (pyrogas) will contain high concentrations of PFAS. Current pyrolysis systems direct primary pyrolysis byproducts, including oil, tar, pyrogas, and exhaust gas, from the thermal dryer to a thermal oxidation unit for PFAS destruction. Natural gas burners supply additional heat to the thermal oxidation unit to raise the gas temperature to more than 1,100°C. Temperatures greater than 1,100°C are required for the defluorination of PFAS.

A portion of the hot exhaust gas is returned to the thermal drying process and to the pyrolysis reactor to recover heat and maintain treatment temperatures. The remaining exhaust gas passes through an exhaust gas scrubber before being released into the atmosphere. A typical thermal oxidizer is depicted in Figure 3-3.

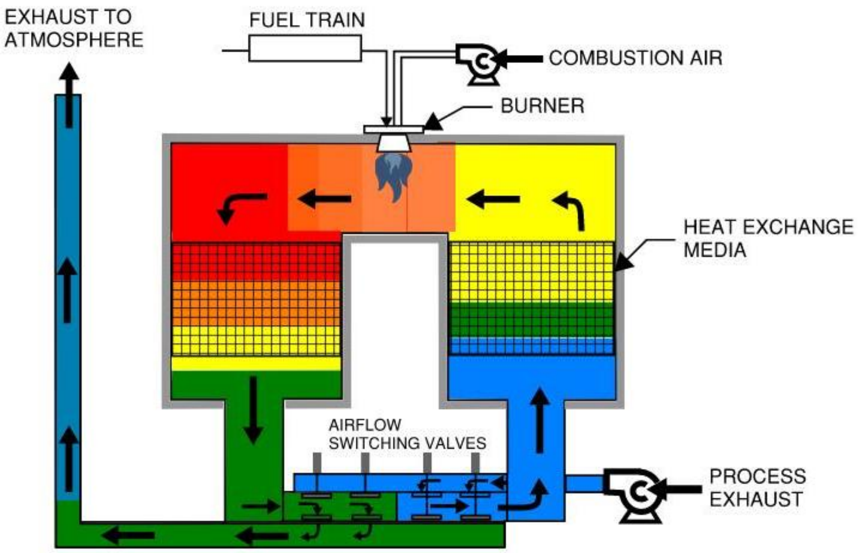


Figure 3-3 Example thermal oxidizer process flow diagram

Thermal oxidation of the gas destroys the PFAS at an efficiency greater than 99% at 1,100°C (Barranco, Caprio, and Hay 2020; Focus Environmental Inc. 2020). A recent pilot study evaluated pyrolysis and subsequent thermal oxidation of pyrogas for the destruction of 41 PFAS and found removal efficiencies between >81.3% and >99.9% (Thoma et al. 2022). However, additional studies are required to determine the PFAS concentration in any solid residuals from high-temperature oxidation of the tars and oils.

Research is ongoing to understand all potential opportunities to use biochar material. Use as a soil amendment, coal substitute in power plants, filtration media, and industrial additive are known opportunities. The biochar may be landfilled if no outlets are identified within a reasonable hauling distance.

3.4.5 Gasification with Thermal Oxidation (retained for disposal of Biosolids)

Gasification is a thermochemical process where pre-dried (<75% moisture) biosolids are heated in a reactor, typically from 675ºC to 815ºC. In gasification systems, a controlled amount of air, oxygen, or steam is added to the reactor to make it an "oxygen-starved," reducing environment, with oxygen levels near 30% of those required for complete oxidation. The gasifier produces PFAS-laden syngas and low-carbon biochar. The temperature conditions and processing time of the biosolids can be controlled to maximize the caloric (heating) value of the syngas. The syngas is then fully oxidized in a separate chamber with excess air for complete combustion (thermal oxidation). Figure 3-4 depicts an example process flow diagram for a gasification system.

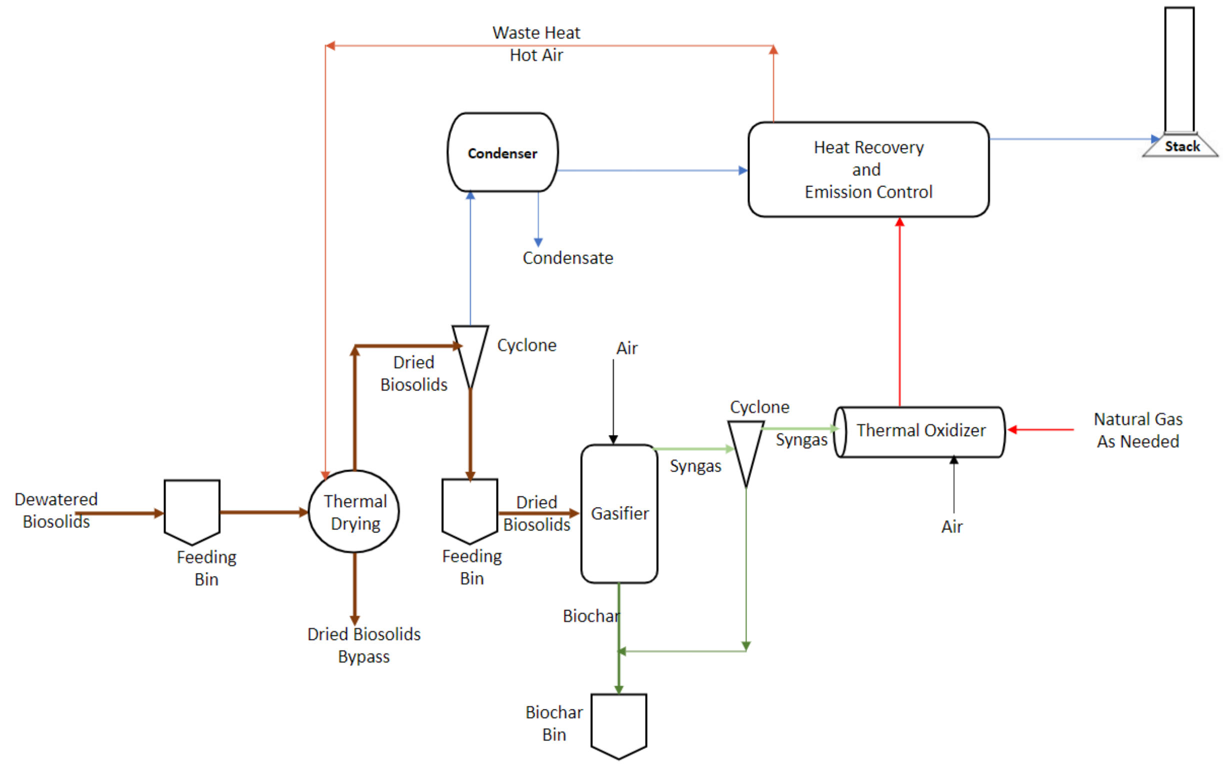


Figure 3-4 Example gasification process flow diagram

Gasification is anticipated to effectively destroy many, but not all, PFAS. Some research indicated limited removal from gasification alone, but 99.9% of the sum of analyzed PFAS was removed between the feed and the resulting char when coupled with thermal oxidation (Davis 2020). Results provided by vendors of full-scale systems showed PFAS removal from waste streams and air phases with the coupled processes. Thermal oxidation is described in more detail in the pyrolysis with the thermal oxidation section, Section 3.4.4.

4 Assembly of Currently Feasible Management Alternatives for PFAS Removal

PFAS management alternatives in this study were compiled from currently feasible treatment technologies identified as effective and scalable when this Report was written (see technology screening described in Section 3). Compiled alternatives were developed using process understanding and previous project experience with the goal of including a range of options likely to perform well in the detailed alternative screening evaluation. In assembling currently feasible PFAS management alternatives, this study included final PFAS destruction options in each alternative. These alternatives are meant to address comprehensive PFAS management at specific facilities. Consideration of regional solutions is included later in Section 10.

4.1 Liquid Waste Streams (Municipal WRRF Effluent, Mixed MSW Landfill Leachate, and Compost Contact Water)

Assembled liquid management alternatives were evaluated for relevance to each liquid waste stream (municipal WRRF effluent, mixed MSW landfill leachate, and compost contact water), as described below. Four to 12 management alternatives were considered and screened for the three liquid waste streams, as summarized in Table 4-1 and further described below.

Technologies selected for **municipal WRRF effluent** needed to be demonstrated at a similar scale to be considered.

- Foam fractionation has not been applied at the 0.1–10 MGD scale assumed here for municipal WRRF effluent, so those alternatives were not considered.
- Modified clay media, specialty wide-spaced RO, and SCWO have also not been applied at a scale relevant to primary treatment without preconcentration using RO but were included in alternatives where those technologies are applied to RO concentrate.
- Alternatives with RO as a pre-concentration step were included (see options with a “b”). This lowers the required size of downstream treatment processes. An added benefit of using RO pre-concentration before GAC adsorption of RO concentrate is a lower GAC media usage rate. This is related to a higher mass of PFAS sorbed per mass of GAC media when fed with higher PFAS concentrations (Franke et al. 2019).

Alternatives selection for **mixed MSW landfill leachate** needed to be suitable for use in liquids with high organics and salt content, even after pretreatment (refer to Section 6.5 for more information).

- Alternatives, including single-use and regenerable AIX, were not evaluated; these are expected to incur significant fouling from salt and organic concentrations expected in mixed MSW landfill leachate.
- Specialty wide-spaced RO has been used to treat landfill leachate and has demonstrated high rejection of PFAS with lower pretreatment requirements than conventional RO. However, this

technology was not incorporated into alternatives because concentrated brine would still require PFAS destruction to meet the goals of this study. RO concentrate water requires additional treatment to remove organics and total suspended solids (TSS) before other separation technologies (GAC and modified clay), so it is not expected to provide an advantage over conventional RO.

For **compost contact water**, the water flow rate was expected to be low, and the overall treatment system complexity will need to align with the ability of the composting facility to construct, operate, and maintain a PFAS water treatment system.

- Alternatives with an RO concentration step (using either conventional RO or specialty wide-spaced RO) were not considered for compost contact water treatment because they were considered too operationally complex for implementation at a compost site.
- Due to the high salt and organic content and potential resin fouling, single-use and regenerable AIX were also not considered for compost contact water.

Table 4-1 Summary of liquid phase PFAS management alternatives

	PFAS Management Alternative (assembled from technologies passing threshold screening and described in Section 3)	Selected for WRRF Effluent Evaluation?	Selected for Mixed MSW Landfill Leachate Evaluation?	Selected for Compost Contact Evaluation?
1a	GAC, high-temperature incineration	X	X	X
1b	RO, GAC, high-temperature incineration	X	X	
2a	Single-use AIX, high-temperature incineration	X		
2b	RO, single-use AIX, high-temperature incineration	X		
3a	Regenerable AIX, AIX regeneration, GAC, high-temperature incineration	X		
3b	RO, regenerable AIX, AIX regeneration, GAC, high-temperature incineration	X		
4a	Regenerable AIX, AIX regeneration, SCWO	X		
4b	RO, regenerable AIX, AIX regeneration, SCWO	X		
5a	Modified clay media, high-temperature incineration		X	X
5b	RO, modified clay media, high-temperature incineration	X	X	
6a	GAC, single-use AIX, high-temperature incineration	X		
6b	RO, GAC, single-use AIX, high-temperature incineration	X		
7b	RO, SCWO[1]	X	X	
8a	Foam fractionation, high-temperature incineration		X	X
8b	Foam fractionation, SCWO		X	X

[1] This alternative is labeled as a “b” option to match with other options, including RO. There is no alternative 7a evaluated because SCWO is likely not economically viable for direct liquid treatment at these scales due to high co-fuel requirements.

4.2 Biosolids

Based on the technology screening, three PFAS management alternatives were identified for municipal WRRF biosolids. Biosolids treatment technologies for PFAS that passed the initial technology screening destroy PFAS as part of the normal operation of the technology. The same process that pyrolyzes or oxidizes the biosolids acts simultaneously to destroy the PFAS compounds. Table 4-2 includes a summary description of the PFAS management alternatives assembled for treating biosolids.

Table 4-2 Summary of PFAS management alternatives for biosolids

	PFAS Management Alternative	Benefit
1	Pyrolysis, thermal oxidation of pyrogas	Produces more biochar, which may be used in a granular media application. Can reuse energy from thermal treatment for thermal drying.
2	Gasification, thermal oxidation of syngas	Produces less biochar, leaving less post-processing material to manage. Can reuse energy from thermal treatment for thermal drying.
3	SCWO	Flexibility to process biosolids with higher water content than pyrolysis or gasification; requires less pre-processing. Can be energy-producing if feed streams have high heating value.

5 PFAS Management Alternatives Evaluation and Cost Curve Methods

5.1 Goals and Approach to Alternative Evaluation Criteria Development and Weighting

The combined PFAS management alternatives identified in Section 4 were scored using a detailed screening matrix of criteria and sub-criteria outlined in Section 5.1.1. Criteria and sub-criteria were assigned numeric weights to reflect their relative importance to each waste stream. The numeric weights were multiplied by the individual scores, and the resulting values for all criteria were summed for each alternative. Higher overall scores correspond to more viable alternatives.

The influent water or biosolids quality and other project assumptions considered for each waste stream are described in Sections 6.1, 7.1, 8.1, and 9.1.

5.1.1 Technical Feasibility

5.1.1.1 PFAS Separation Efficiency (Ability to Meet 5 ng/L or 5 ng/g)

This screening criterion was intended to evaluate each alternative by the ability to meet the PFAS targets in any residual liquid that would subsequently be discharged to the environment, sanitary sewer, or reused (treated water). The ability of an alternative to separate PFAS was based on the potential to achieve the treated water quality targets provided by the MPCA based on published results from similar applications. Alternatives that received the highest scores have PFAS concentrations in treated liquid streams below 5 ng/L for liquids and 5 ng/g for biosolids for all target PFAS listed in Table 2-1. Treated liquid streams comprise sorption media vessel effluent for alternatives using media sorption, blended RO permeate and media vessel effluent for alternatives using both, and RO permeate for alternatives where RO is the only separation technology.

- Scores correspond with the following separation abilities:
  - 1 - <= Treated water <5 ng/L for one or two target PFAS
  - 2 - <= Treated water <5 ng/L for over half of target PFAS
  - 3 - <= Treated water <5 ng/L for all target PFAS

While some technologies report a percent separation or percent destruction, others evaluated in this study separate or destroy nearly all PFAS present—at least for a defined operating period. For example, GAC adsorption can achieve near-complete separation of PFAS until sorption media is saturated or sorption kinetics results in desorption of previously sorbed PFAS. As another example, pyrolysis of biosolids coupled with thermal oxidation achieves near-complete destruction of PFAS.

For media-based separation, scoring considered treated effluent assuming consistent changeout frequency and equipment sizing for typical hydraulic loading rates and empty-bed contact times (e.g., 10–

20 minutes per bed for GAC, 2–10 minutes per bed for modified clay media, and 2–6 minutes per bed for AIX resin). This was intended to capture differences in breakthrough periods and sorption media use rates for different media types. This criterion is not applicable to biosolids.

5.1.1.2 PFAS Destruction Efficiency

This screening criterion was intended to evaluate the PFAS destruction efficiency of each alternative. The ability of an alternative to destroy PFAS was based on the reported removal of target PFAS provided by the MPCA. Alternatives that received the highest scores have demonstrated PFAS mineralization across the process for target PFAS. All destruction technologies that passed the threshold screening can potentially create additional PFAS as byproducts, with more information needed on total organic fluorine mass balances. Thus, the scoring for this criterion was differentiated by measured removal of target PFAS only.

- Scores correspond with the following destruction abilities:
    - 1 - >= 90% reported removal of target PFAS
    - 2 - >= 99% reported removal of target PFAS
    - 3 - >= 99.9% reported removal of target PFAS

5.1.1.3 Degree of Commercialization

Alternatives that include PFAS separation and destruction technologies with a range of commercialization were evaluated using multiple feasible configurations. Alternatives containing technologies with a higher composite degree of commercialization were ranked higher for this criterion.

- Scores correspond with the following degrees of commercialization for the primary separation or destruction technology for liquid waste streams:
    - 1 - Commercial application of the primary separation and destruction technology in one industry each
    - 2 - Commercial application of the primary separation and destruction technology in two-to-three industries each
    - 3 - Commercial application of the primary separation or destruction technology in more than three industries each, including municipal wastewater treatment and/or landfill leachate treatment
- Scores correspond with the following degrees of commercialization for biosolids technologies:
    - 1 - Pilot study of the destruction technology for biosolids
    - 2 – Two commercial applications of the destruction technology for non-biosolids applications with a pilot study in progress or completed for biosolids

    - 3 – Three or more commercial applications of the destruction technology for biosolids

5.1.1.4 Reliability of Performance

Reliability of performance applies to the overall ability of a PFAS management alternative to produce treated effluent or biosolids that consistently meet the PFAS treatment targets given variable influent liquid or biosolids quality, variable flows or loading rates, or variable environmental conditions.

- Scores correspond with the following qualitative level of performance reliability for the alternative as a whole:
    - 1 - Reliability may be inconsistent or unknown
    - 2 – Intermediate reliability
    - 3 - Most reliable

5.1.1.5 Simplicity of Operation and Maintenance

The relative simplicity of operation and maintenance requirements for each alternative was evaluated based on the number of process steps, pre- and post-primary separation or destruction technology requirements, level of operator attention required, level of operator certification/training required, and the anticipated number of operational changes required under normal operating conditions.

- Scores correspond with the following level of complexity for the alternative as a whole:
    - 1 - Most complex
    - 2 - Average complexity
    - 3 - Most simple

5.1.1.6 Operator and Public Health

The operator and public health criterion refers to known health risks to the individual operators and the general public associated with the operation of each alternative. This includes direct and indirect exposure risks attributable to PFAS and other potentially harmful substances. It also includes more general safety risks associated with equipment operation and effluent and byproducts management.

- Scores correspond with the following level of complexity for the alternative as a whole:
    - 1 - Moderate additional health risk
    - 2 - Some additional health risk
    - 3 - Limited additional health risk

5.1.2 Economic Feasibility

Five unique economic criteria were used to compare the anticipated costs of each alternative based on available information from existing applications and technology vendors. Economic feasibility criteria included the following.

5.1.2.1 Relative Capital Costs

Under this criterion, lower rankings were given to the most expensive alternatives and higher rankings to relatively low-cost alternatives. Alternatives incorporating fewer unit processes and less mechanical equipment generally had lower capital costs. Capital cost screening was based on relative costs within the four specific waste streams targeted in this study.

- Scores correspond with the following for the alternative as a whole:
    - 1 - High relative capital cost
    - 2 - Average relative capital cost
    - 3 - Low relative capital cost

5.1.2.2 Relative Operation and Maintenance Costs (Not Including Energy Costs)

O&M costs include labor, parts and maintenance, chemicals for operations and cleaning processes, consumables, and byproducts management. Energy usage (power) was considered separately from other O&M costs. Lower rankings were given to the alternatives with the highest relative O&M requirements/costs and higher rankings to relatively low O&M demand alternatives. O&M cost screening was based on relative costs within the four specific waste streams targeted in this study.

- Scores correspond with the following for the alternative as a whole:
    - 1 - High relative O&M cost
    - 2 - Average relative O&M cost
    - 3 - Low relative O&M cost

5.1.2.3 Relative Energy Consumption

Under this criterion, lower rankings were given to the alternatives with the highest energy consumption and higher rankings to relatively low-energy demand alternatives. Alternatives incorporating unit processes with high-temperature requirements for PFAS destruction had the highest energy consumption.

- Scores correspond with the following for the alternative as a whole:
    - 1 - High relative energy consumption
    - 2 - Average relative energy consumption

  - 3 - Low relative energy consumption

5.1.2.4 Relative Complexity and Cost of Pretreatment

Under this criterion, lower rankings were given to the alternatives requiring pretreatment ahead of PFAS treatment. This includes alternatives with sorption media and RO membrane separation technologies. Liquid waste stream alternatives that use foam fractionation were not expected to require pretreatment beyond a simple particulate filter, thus simplifying implementation.

- Scores correspond with the following for the alternative as a whole:
  - 1 - High relative pretreatment complexity
  - 2 - Average relative pretreatment complexity
  - 3 - Low relative pretreatment complexity

5.1.2.5 Relative Energy Recovery from Biosolids Alternatives

The thermal technologies used to desorb and destroy PFAS from biosolids can recover energy from the heat remaining in the process after meeting the energy demand of the process itself. Heat energy can be used to supplement building or process heating systems or to generate electric power. This criterion is separated from energy consumption in this evaluation because energy recovery can potentially affect the favorability of some alternatives. Under this criterion, the lowest rankings were given to the alternatives with no recoverable energy. Alternatives that incorporated unit processes with ample recoverable energy were scored the highest.

- Scores correspond with the following for the alternative as a whole:
  - 1 - No recoverable energy from biosolids process
  - 2 - Low relative recoverable energy from biosolids process
  - 3 - High relative recoverable energy from biosolids process

5.1.2.6 Applicability at Scale

This criterion refers to the scale at which alternatives have been implemented in field and full-scale applications, as well as considerations for the commercial availability of equipment and supplies. It also considers economies of scale at the design basis sizing for each waste stream. While primary separation or destruction technologies may be commercialized, these applications may be at a scale much smaller than required for a specific waste stream in this study.

- Scores correspond with the following for the alternative as a whole:
  - 1 – Minimal commercial application at a relevant scale (0–2 applications)
  - 2 – Limited commercial application at a relevant scale (2–10 applications)

    - 3 – Significant commercial application at a relevant scale (>10 applications)

Other considerations that affect applicability at different scales include the availability of sorption media, chemicals, and other materials needed for treatment. These were not considered in this evaluation.

5.1.3 Byproducts Management

Byproducts include treated water (post-primary separation or destruction), spent sorption media, waste streams, combustion-related solids/ash, etc. Consideration of byproducts management is divided into two criteria to evaluate the potential for byproduct reuse after PFAS have been removed or destroyed as well as the potential for air emissions of PFAS or other air pollutants not currently measured.

5.1.3.1 Beneficial Reuse Opportunity for Water or Byproducts

This criterion applies to the potential use of PFAS separation and destruction byproducts, such as water reuse instead of discharge and GAC reactivation. Beneficial reuse of treated water or solid byproducts may offset PFAS separation and destruction costs, though this is site-specific. In addition, high-quality treated water could be used for irrigation, industrial reuse, and indirect potable reuse applications, potentially offsetting water treatment costs in the region.

- Scores correspond with the following for the alternative as a whole:
    - 1 – No beneficial reuse of water or solid byproducts
    - 2 – Potential beneficial reuse of water or solid byproducts
    - 3 – Established beneficial reuse of water or solid byproducts

5.1.3.2 Potential for Media Shifting of PFAS Pollutants

The potential for media shifting of PFAS refers to the composition and disposition of residual materials and how likely remaining PFAS are to partition into another phase (pathways such as air, ash, etc.) during management, reuse, destruction, or disposal. Significant uncertainty remains regarding the loss of PFAS, including compounds not included in current analytical methods that could be volatilized or formed during existing and developing thermal destruction processes.

- Scores regarding qualitative byproduct transformation potential correspond with the following for the alternative as a whole:
    - 1 – High media shifting potential, likely development of PFAS byproducts of incomplete combustion
    - 2 – Possible media shifting potential; unknown possibility of development of PFAS byproducts of incomplete combustion
    - 3 – Low media shifting potential, unlikely development of PFAS byproducts of incomplete combustion

5.1.4 Evaluation Criteria Selection and Weighting by Waste Stream

Criteria weighting differed by waste stream depending on specific priorities for each industry. Scoring of each alternative for each criterion also varied among media depending on the scale, pretreated water quality, and other site considerations associated with each waste stream evaluated.

Table 5-1 summarizes the criteria weighting used for different waste streams. Each criterion was given a weight of one through five, which was multiplied by the one-to-three score for each alternative for a weighted score. Some of the considerations used in the development of criteria weighting included the following:

- Municipal WRRF effluent and biosolids treatment alternatives require more commercialization and reliability than alternatives applied to landfill leachate and compost contact water, partly due to the larger volume of material requiring management and partly due to the importance of consistent wastewater treatment to society.
- Beneficial reuse opportunities for biosolids treatment alternatives are important to decrease or eliminate the need for landfill disposal of the residuals.
- Pretreatment complexity is more important for lower-quality liquid streams like mixed MSW landfill leachate and compost contact water than municipal WRRF effluent. The lack of existing treatment translates to a higher operational burden imposed by extra treatment processes. All biosolids management options require pretreatment via dewatering, so there was no difference in complexity and cost among alternatives.

Table 5-1 Evaluation criteria weightings by waste stream

Category	Municipal WRRF Effluent	Biosolids	Mixed MSW Landfill Leachate	Compost Contact Water
Technical Feasibility				
PFAS separation efficiency	5	N/A	5	5
PFAS destruction efficiency	5	5	5	5
Degree of commercialization	4	4	3	3
Reliability of performance	3	4	2	2
Simplicity of operation/maintenance	3	2	4	4
Operator and public health	2	2	2	2
Economic Feasibility				
Relative capital costs	4	4	4	4
Relative operation and maintenance costs	3	3	3	3
Relative energy consumption	2	2	2	2
Relative complexity and cost of pretreatment	2	1	4	4
Energy recovery options (applicable for biosolids technologies only)	N/A	2	N/A	N/A
Applicability at scale	5	5	5	5
Byproducts Management				
Beneficial reuse opportunity for water or byproducts	1	3	1	1
Potential for media shifting of PFAS	2	2	2	2
Minimum score achievable[1]	**41**	**39**	**42**	**42**
Maximum score achievable[1]	**123**	**117**	**126**	**126**

[1] The minimum score achievable by a given alternative is the sum of criteria weights multiplied by one (the lowest allowable score for each criterion). The maximum score achievable is the sum of criteria weights multiplied by three (the highest allowable score for each criterion).

5.2 Goals and Approach for Preliminary Design and Costing of Example Pretreatment Facilities

Some PFAS management alternatives require the pretreatment of municipal wastewater, landfill leachate, or compost contact water before entering PFAS separation processes. Alternatives that require pretreatment include RO membrane separation or sorption media pressure vessels (with GAC, AIX, or modified clay media). Other alternatives for liquid waste streams that rely on foam fractionation for PFAS separation or destruction were not expected to require pretreatment.

Pretreatment requirements and design basis for liquid waste streams are expected to vary considerably among specific sites. This study outlines pretreatment water quality requirements for liquid waste streams and anticipated influent water quality for PFAS management alternatives. It then outlines an example pretreatment process flow that could meet pretreatment water quality requirements in some instances and associated costs, as provided in Sections 6.5, 8.5, and 9.5. These evaluations were meant to provide an approximate order-of-magnitude cost estimate for pretreatment and should not replace site-specific treatability evaluations for select pretreatment processes. Because the focus of this study is PFAS separation and destruction and pretreatment requirements were expected to vary significantly by site, pretreatment cost estimates were developed in less detail, with greater uncertainty (at +100%/-50%) than PFAS separation and destruction cost estimates (at +50%/-30%).

To optimize the treatment efficacy of the PFAS management alternatives, it is recommended that the influent water quality for the aqueous waste streams meet the pretreated concentrations outlined in Table 5-2. These pretreatment concentrations are based on vendor-recommended influent water quality for RO membrane separation, GAC sorption, and AIX technologies.

Table 5-2 Targeted pretreated water quality for PFAS management alternatives with RO or sorption media vessels

Parameter[1]	Units	GAC Influent Targets	AIX Influent Targets	RO Influent Targets	Rationale
TOC[2]	mg/L	20	2	2	Membrane and sorption media foulant
TSS	mg/L	1	1	1	Membrane and sorption media foulant
TDS	mg/L	NA	500	NA	AIX foulant; affects RO recovery
Iron	mg/L	0.5	0.1	0.1	Limits fouling
Manganese	mg/L	0.5	0.1	0.1	Limits fouling

[1] Pretreatment thresholds for these parameters were not established for modified clay media. However, it is assumed that modified clay media must achieve similar pretreatment targets as GAC to avoid media fouling and pore blockage issues.

[2] Dissolved organic carbon (DOC) is the parameter directly affecting fouling. However, TOC is used here because it is more commonly monitored and because the low TSS means that DOC and TOC concentrations are very similar. TOC is listed as the primary organic bulk parameter, but if not measured, BOD_5 should be less than 60 mg/L.

PFAS destruction alternatives for biosolids require pretreatment to remove material that could damage the destruction equipment and dewater the biosolids enough to meet SCWO, pyrolysis, or gasification feed requirements. The PFAS destruction alternatives' solids concentration requirements for biosolids are outlined with associated costs in Section 7.5.

5.3 Goals and Approach for Preliminary Design and Costing of Site-Specific Facilities

Alternatives that scored the highest in the detailed screening for each waste stream were carried forward for preliminary design and cost-curve development. This includes four alternatives for municipal WRRF effluent, two for municipal WRRF biosolids, four for mixed MSW landfill leachate, and three for compost contact water.

The following components for each site-specific PFAS management alternative can be found in Sections 6 through 9:

- Process flow diagram
- Design basis summary table
- Class 5—Association for the Advancement of Cost Engineering International Recommended Practice No. 17R-97 (AACE International 2020) capital cost table and scalable cost curve
- O&M cost table and scalable cost curve
- Narrative summarizing any relevant implementation concerns related to design, construction, operation, or maintenance for each alternative developed

5.3.1 Preliminary Design

Preliminary design basis summary tables in subsequent sections summarize key design parameters selected for each unit process comprising a single alternative.

The purpose of the preliminary design was to develop equipment needs and implementation considerations and support Class 5 (AACE) cost estimating. These preliminary designs were developed based on conversations with vendors, as well as relevant project experience and professional judgment. Preliminary designs were developed to address the design basis, including PFAS management alternative influent water and biosolids quality. These are outlined in subsequent sections for each targeted waste stream and are not intended to be applied to facilities with different conditions. The actual feasibility and effectiveness of any technology can vary from that assumed for a variety of site-specific reasons. Bench-scale or pilot-scale testing (or further testing, etc.) will be needed if more certainty as to the effectiveness of the technology based on site-specific conditions is required.

Specific design basis decisions that affect multiple waste streams include the following:

- 15-minute EBCT for GAC sorption media, reflecting a value slightly higher than the industry standard of 10 minutes for PFAS removal due to relatively low PFAS treatment targets.
- 4-minute EBCT for AIX sorption media, reflecting a value slightly higher than the industry standard of 2 minutes for PFAS removal due to relatively low PFAS treatment targets.

- 10-minute EBCT for modified clay sorption media, based on vendor feedback that EBCT should be higher than for AIX for high-PFAS waste streams like mixed MSW landfill leachate and compost contact water.

- Target hydraulic loading rate of 1–10 gpm/ft^2 for GAC vessels and 5–18 gpm/ft^2 for AIX and modified clay vessels.

- RO flux of 16 gallons per square foot per day.

- RO recovery of 85% for tertiary wastewater effluent and 65% for pretreated mixed MSW landfill leachate.

5.3.2 Cost Estimate Approach and Limitations

A Class 5 (AACE) capital cost estimate was developed for each PFAS management alternative carried forward to a preliminary design. Each cost estimate was then scaled to flows within the design basis flow range for each waste stream to develop cost curves. Capital costs are presented in summary tables for three flow rates (low, middle, and high) for each waste stream and management alternative, with cost curves spanning the design basis flow range. Each capital cost curve has an estimated accuracy within a margin of +50%/-30%, consistent with less than 2% maturity of project design deliverables (AACE International 2020).

Sources used in the development of the capital cost curves included existing cost estimates prepared by Barr and Hazen as part of recent economic evaluations for multiple PFAS treatment and destruction alternatives, Barr's cost estimate repository, Minnesota project experience, conversations and budgetary estimates from vendors, and resources available through Barr's membership and involvement with the Interstate Technology Regulatory Council PFAS Team.

O&M costs were also developed based on the preliminary designs for the PFAS management alternatives and capital cost estimates. O&M costs also relied on vendor input, relevant project experience, and current industry rates for labor, electrical, sorption media, transportation, and disposal. The O&M costs are assumed to have the same level of accuracy (+50%/-30%) as the capital cost estimates.

The capital and O&M cost estimates developed for this Report represent the best judgment of the experienced and qualified professionals familiar with this project. Using the above sources, the estimated capital and O&M costs developed for preliminary PFAS management alternatives in this Report are intended to compare alternatives rather than represent estimates for specific projects. These cost estimates should not be directly applied in situations with different water or solids quality, operational constraints, or site-specific conditions. Barr has no control over the cost of labor, materials, equipment, or services furnished by others, over any contractor's methods of determining prices, or over competitive bidding or market conditions. Barr expects that costs for a specific site will vary from the opinions of probable capital and O&M costs for preliminary PFAS management alternatives developed for this Report. If managers of specific sites wish greater assurance as to probable costs, they should conduct a site-specific feasibility evaluation, including pilot testing, to characterize site-specific considerations and needs and to more accurately estimate PFAS management costs.

5.3.3 Capital Cost Support

PFAS separation and destruction equipment costs were solicited from multiple vendors who were presented with representative pretreated liquid waste stream or biosolids quality and characteristics and flow rates. Vendors were asked to assemble equipment options, footprints and dimensional diagrams, and capital costs. The information received from vendors was evaluated and compared for selecting representative equipment capital costs for inclusion in the preliminary cost estimates for each alternative at each flow rate evaluated. Specific vendor equipment models and the breakdown of equipment costs between unit processes, where an alternative includes multiple units, are not reported in the design basis or capital cost estimate tables. For most alternatives, the most conservative equipment costs were selected for inclusion at this preliminary design stage.

Equipment sizing for sorption vessels treating RO concentrate was accomplished by estimating achievable RO recovery. Lower recovery rates reduce energy use and fouling rate but lower permeate flux. The water quality and the required percentage removal of some of the water constituents also affect potential operating conditions and target recovery of RO membranes. For the current work, the recovery rate for the RO membranes treating municipal WRRF effluent and mixed MSW landfill leachate was set at 85% and 65%, respectively, based on literature and water quality.

For alternatives that include multiple unit processes, the total equipment cost was reported as a single line item in the capital cost estimates. Equipment costs were marked up by 28% to account for the estimated contractor markup (20%), and the average Minnesota sales tax was rounded up to the nearest percentage point (8%).

Most of the remaining capital-cost estimate line items were calculated as percentages of equipment, building, and/or construction costs based on relevant project experience and considering the level of preliminary design, except for treatment building costs. The percentages and line-item inclusions were as follows:

- General conditions: 15% of construction and equipment subtotal
    - Includes: Contractor mobilization and demobilization and site and project management
- Piping and appurtenances: 50% of total equipment cost
    - Includes: Separation and destruction equipment interconnected piping, piping supports, valves, and fittings
- Electrical and instrumentation/controls: 20% of total equipment cost
    - Includes: Controllers, valves, switches, indicators, transmitters, converters, instruments, conduits, and supports
- Site work: 10% of building cost
    - Includes: Yard piping and utilities

- Installation (equipment and piping): 50% of the total equipment and piping and appurtenances costs
    - Includes: Contractor installation of separation and destruction equipment, process piping and appurtenances, and yard piping and utilities

A treatment building cost of $500 per square foot was assumed, which includes subgrade excavation and preparation, a concrete pad, building materials, heat, power and electrical, mechanical/HVAC/plumbing, and appurtenances. Separation and destruction equipment footprints were estimated with vendor-provided information and dimensional diagrams and rounded to the nearest 10 or 100 square feet, depending on the size of the facility. For alternatives that include media sorption processes, a footprint for a driving lane for media changeout and a pipe gallery were included for liquid waste stream flow rates greater than or equal to 100 gpm (0.14 MGD). For all alternatives, a footprint for a lab and bathroom was included for liquid waste stream flow rates greater than or equal to 500 gpm (0.72 MGD).

With the framework described in this section, capital cost estimate tables were developed for three flow values for each waste stream, targeting each order of magnitude. Where appropriate, cost curves were developed by fitting a power formula to the three costs and flows. In some cases (the very low flow range used for mixed MSW landfill leachate and compost contact water and alternatives with significant modular equipment costs), a linear formula was used to fit the cost curves.

5.3.4 Operation and Maintenance Cost Support

Operation and maintenance cost estimates were developed with the following line items (as applicable):

- Utilities (including energy use for process and non-process loads)
- Sorption media replacement (including media purchase and trucking)
- Sorption media disposal (including spent media disposal and trucking)
- GAC reactivation (applicable to systems with 80,000 pounds or more of spent GAC per changeout event)
- Concentrate disposal (including concentrate disposal and trucking)
- Chemical usage (including flocculants, caustic, antiscalants, and clean-in-place chemicals, as needed)
- Maintenance (including process equipment and general building maintenance)
- Monitoring (including shipping and analytical fees for PFAS monitoring)
- Labor (including payroll and benefits for full-time equivalents for operation and maintenance staff)

For alternatives that include GAC adsorption, thermal reactivation was expected to be more cost-effective than single-use GAC. However, this technology has only been demonstrated as effective for PFAS removal when a facility can designate dedicated (site-dedicated) carbon to be reused only at that site (Westreich et al. 2018). Vendors have advised that the GAC use rates of smaller facilities (e.g., less than 80,000 pounds total per changeout event) are likely too small to enroll in site-dedicated reactivation programs, largely due to constraints on reactivation facility sizes. As a result, alternatives with GAC adsorption were assumed to use thermal reactivation for sites with over 80,000 pounds of GAC per changeout, which only applies to municipal WRRF effluent greater than 1.0 MGD. Other waste streams and smaller WRRF sizes with GAC adsorption were assumed to use single-use GAC with off-site high-temperature incineration. Disposal of GAC via general pool reactivation through a GAC vendor could be a cost-saving option over high-temperature incineration. However, the availability of this option is subject to media acceptance into the general pool reactivation program by the GAC vendor based on the media characteristics. Because of the uncertainty of the availability of this disposal option, it was not considered in the O&M estimates prepared for this study.

Utility use estimates were based on guidelines from the Electric Power Research Institute (Electric Power Research Institute 2013), assuming electrical costs of \$0.13/kW-hr based on commercial pricing in Minnesota in 2022 (U.S. EIA 2022a). The natural gas estimate of \$15.91/kSCF (kSCF=1,000 standard cubic feet) was based on reported commercial pricing in Minnesota in 2022 (U.S. EIA 2022b).

GAC media replacement frequencies were approximated by modeling the estimated bed volumes treated to breakthrough at 5 ng/L. Bed volumes treated to breakthrough using GAC (Calgon F400) were estimated using an advection-dispersion model coupled with a homogenous surface diffusion model (HSDM) (Crittenden et al. 1986). Freundlich isotherm parameters for specific PFAS were taken from Burkhardt et al. (2022). Water-film mass transfer coefficients and surface diffusivities of PFAS on GAC were taken from Jarvie et al. (2005), and water-phase diffusion coefficients for individual PFAS were taken from Schaefer et al. (2019). This modeling exercise assumed influent PFAS concentrations matching those typical for each waste stream: municipal WRRF effluent, mixed MSW landfill leachate, compost contact water, and concentrate from RO treatment of municipal WRRF effluent and mixed MSW landfill leachate. These influent concentrations are presented in waste stream-specific sections in Table 6-1, Table 7-1, Table 8-1, and Table 9-1. PFAS concentrations in RO concentrate were estimated assuming 100% PFAS rejection and 85% water recovery for wastewater effluent and 65% recovery for mixed MSW landfill leachate. Additional details are included in Appendix D.

Mechanistic modeling was not completed for media replacement frequencies of AIX resin and modified clay due to limited data and information available in peer-reviewed literature to enable the development and implementation of mechanistic breakthrough models for these media. Instead, media replacement frequencies were estimated, assuming both AIX resin and modified clay media would have the same volume throughput of treated water (in gallons) as estimated for GAC using the HSDM model. For this study, breakthrough of the short-chain PFAS, PFBA, and PFBS dictated media exchange frequencies for GAC. Thus, for AIX resin beds and modified clay media beds designed to typical industry EBCTs (e.g., 4 minutes and 10 minutes, respectively), it was assumed that each media (GAC, AIX, and modified clay) would achieve the same volume throughput based on the breakthrough of the short-chain PFAS, PFBA or

PFBS. This assumption was supported by similar ranges in the bed volumes to breakthrough for each media for these PFAS as summarized in Appendix C.

Due to the uncertainties in the estimated media replacement frequencies used in this study, Barr recommends that sites seeking to implement media sorption for PFAS removal conduct pilot studies to accurately predict media replacement frequencies based on site-specific water quality and PFAS removal goals.

Costs for sorption media replacement were based on vendor feedback and pricing estimates for the selected sorption media. Costs for high-temperature incineration (for sorption media and foam fractionation foamate) were based on vendor discussions and data summarized in U.S. EPA (2020). GAC reactivation costs were based on vendor discussions. Disposal fees will vary depending on the specific characteristics of the waste and the waste manager. Therefore, the costs presented in this study may not reflect actual costs for the disposal of PFAS wastes, byproducts, or concentrates.

Trucking costs to incineration and reactivation facilities assumed $3/mile (assuming the PFAS-containing wastes are considered non-hazardous), 760 miles to a commercial incinerator that accepts PFAS waste, and 900 miles to a commercial GAC reactivation facility. Mileage was based on approximate distances between the Minneapolis-Saint Paul metro area of Minnesota and a conceptual commercial facility for each type.

Other line items included chemical usage for foam fractionation (cationic surfactant) and SCWO (caustic), antiscalant and clean-in-place chemicals for RO membrane separation, bag filter replacement for foam fractionation, and monthly PFAS sample collection and analysis.

Annual process equipment maintenance costs were assumed to be 3% of the purchase price of the process equipment. Annual general building maintenance was assumed to be $2.50/sq ft.

Labor (operator and maintenance staff time) was adjusted based on the facility size and complexity of the unit processes.

5.4 Goals and Approach for Regional Facility Evaluation

In addition to the site-specific alternatives evaluated, the feasibility and cost-effectiveness of regional PFAS destruction technologies were evaluated. This included a concept-level design for two regional PFAS destruction facilities and capital and O&M cost estimates for each. Operating costs (or savings) unique to regional facilities were developed to establish a potential payback period compared to discrete treatment systems.

The following components, outlined in Section 10, were developed for each regional PFAS destruction facility:

- Process flow diagrams
- Capital cost estimates

- O&M cost estimates and payback evaluation
- Narrative summarizing any relevant implementation concerns related to design, construction, operation, or maintenance for each alternative developed

The limitations of the PFAS management cost estimates outlined in Section 5.3 also apply to cost estimates for regional PFAS destruction facilities described in Section 10. Capital costs were based on a combination of vendor quotes for available equipment (for a regional biosolids pyrolysis facility) and scaling from recent reports of costs of facility expansions (for a regional high-temperature sorption media incineration facility).

Net present value (NPV) and payback period were used to evaluate the potential economic feasibility of new regional facilities. NPV reflects the current value of the net return at the end of the planning period relative to what might have been gained by investing elsewhere. The interest rate was assumed to be 2–6% (achievable market rate interest less inflation). This reflects the minimum attractive rate of return that could be achieved by investing capital elsewhere. A range was used to reflect a high degree of future uncertainty due to currently high inflation rates. The payback period is the time the capital expense can be repaid from the project benefits, not considering interest.

The effects of secondary environmental impacts (externalities) were also estimated for each regional facility based on literature regarding the cost of these impacts.

6 Municipal Wastewater

6.1 Project Assumptions for Municipal Wastewater

6.1.1 Description and Scale of Required Treatment

In the United States, centralized municipal WRRFs are the primary means by which wastewater is treated to remove biochemical oxygen demand, nutrients, and pathogens prior to discharge back to the environment. The majority of water treated comes from municipal sources, which contain PFAS due to their wide use in everyday consumer products (Thompson et al. 2022; Roy et al. 2018; Hamid, Li, and Grace 2018; Choi et al. 2019). Many WRRFs also accept industrial wastewater discharges under pretreatment agreements, which have the potential to contribute additional PFAS loading to WRRFs (Thompson et al. 2022). Some WRRFs serve communities with PFAS-contaminated potable water above the treatment targets identified in this study, and could thus exceed the treatment targets without additional PFAS loading from municipal or industrial sources (McNamara et al. 2018; Appleman et al. 2014; Pan, Liu, and Ying 2016). For this study, high industrial loads of PFAS were assumed to be managed via industrial pretreatment agreements, whereby PFAS are removed from industrial discharges before entering a WRRF. The concentrations assumed as design basis reflect typical values for Minnesota facilities without significant industrial PFAS loading. While this assumption is not realistic for every Minnesota WRRF, given current source management practices, it enables the development of baseline treatment needs and costs that can be increased, if needed, for facilities with significant industrial PFAS inputs.

Characteristics for two different types of existing, conventional treatment were used to reflect the majority of WRRFs in Minnesota:

- Conventional activated sludge treatment, including grit removal, primary sedimentation, secondary activated sludge with nitrification, and disinfection
- Stabilization pond treatment, including treatment in unaerated stabilization ponds, followed by disinfection

PFAS treatment of WRRF effluent would be applied downstream of existing treatment (end-of-pipe) to provide high-quality water that reduces fouling and maintenance costs associated with removing PFAS. This study addressed and produced cost curves for removing and destroying PFAS from WRRFs with influent flows ranging from 0.1 to 10 MGD and, as noted, assumes no significant industrial contributions of PFAS.

While the primary goal of the study was to evaluate PFAS treatment alternatives for their ability to separate and mineralize PFAS, WRRFs may be interested in the ability of retrofit equipment to meet other water quality goals, including removing the following:

- Total nitrogen to less than 7 mg/L
- Total phosphorus to less than 0.5 mg/L

- Total mercury to less than 6.9 ng/L

Pretreatment retrofits achieving tertiary treatment are expected to be needed for all PFAS management alternatives evaluated in detail and are described in more detail in Section 6.5. This tertiary treatment is expected to meet the above water quality goals, so the ability of PFAS management alternatives to do so was not evaluated. An added benefit of tertiary treatment is the potential for beneficial reuse, including possibly industrial and irrigation uses as well as indirect potable reuse.

6.1.2 PFAS Influent Concentrations and Treatment Targets

The presence and abundance of specific PFAS in WRRF influent are variable and reflective of ongoing consumer and industry changes. The PFAS most relevant in municipal wastewaters include short-chain PFCAs, long-chain legacy PFCAs and PFSAs, and fluorotelomers, with target PFAS selected as previously described in Section 2.3. Of the PFAS mass entering a WRRF, long-chain PFAS preferentially sorb to biosolids, and short-chain PFAS preferentially remains in the liquid phase (Guo et al. 2010; Pan, Liu, and Ying 2016). However, the transformation of PFAS precursors within the WRRF can lead to an increase in PFAA concentrations between WRRF influent and WRRF effluent (Thompson et al. 2022; Houtz et al. 2016).

Assumed typical and high WRRF effluent PFAS concentrations for use in this study were developed based on MPCA data for Minnesota WRRFs and are summarized in Table 6-1. Typical and high values were selected to reflect approximate median and 90th percentile concentrations observed in the source data and serve as a basis for developing design basis and cost estimates in this study. These values are not intended to reflect the full range of PFAS concentrations that may be present at WRRFs.

Table 6-1 Assumed influent concentrations and treatment goals for target PFAS in municipal WRRF effluent (all units in ng/L)

PFAS[1]	Typical Concentrations[2]	High Concentrations[2]	Treatment Target
PFBA	15	200	5
PFBS	15	50	5
PFHxA	30	60	5
PFHxS	5	15	5
PFOA	40	80	5
PFOS	5	30	5
6:2 FTS	5	150	5

[1] Total oxidizable precursor assay is identified as an important analysis for municipal wastewater, but representative typical and high numerical concentrations were not available from the referenced datasets.

[2] Typical concentrations were selected to be in the range observed in three main sources: 2009–2010 MPCA-supplied data; Helmer, Reeves, and Cassidy (2022); and Coggan et al. (2019). Data reflect aqueous concentrations in treated effluent from municipal WRRFs.

6.1.3 Water Quality Assumptions for Pretreated WRRF Effluent

Non-PFAS water quality parameters have significant implications for currently available PFAS removal technologies. This section describes general water quality assumptions for municipal WRRF effluent. Representative influent and effluent water quality reflecting a typical WRRF in Minnesota was developed for non-PFAS parameters and listed in Table 6-2. This water quality includes moderate-to-high hardness of 511 mg/L as $CaCO_3$ and high alkalinity of 410 mg/L as $CaCO_3$. While this water quality will not be accurate for any one facility, it serves as a starting point to enable preliminary design and cost estimating for Minnesota WRRFs.

Effluent from conventional activated sludge or stabilization pond WRRFs is expected to contain concentrations of suspended solids and organic matter that would cause operational issues with primary PFAS separation technologies (e.g., GAC adsorption, AIX, and RO membrane separation). As a result, pretreatment consistent with tertiary treatment technologies is needed. Table 6-2 also includes tertiary-treated water quality used as the design basis for preliminary design and cost estimates. This water quality is assumed to be the influent to PFAS separation technologies and was based on WRRF effluent water quality and pretreatment targets for media sorption (GAC and AIX) and RO processes discussed in Section 5.2. Examples of WRRF retrofits and potential costs to provide this water quality prior to PFAS separation are further discussed in Section 6.5.

Table 6-2 Assumed initial and pretreated municipal wastewater quality

Parameter	Units	Municipal WRRF Influent[1]	Concentration in Mechanical Secondary Effluent[1,2]	Assumed Concentration in Pretreated Water (Influent for PFAS Management Alts)
pH	SU	Not reported	Not reported	7[3]
TDS	mg/L	Not reported	Not reported	500[3]
TSS	mg/L	200	5	1[4]
BOD_5	mg/L	200	8	2
TOC	mg/L	Not reported	Not reported	2[4]
Alkalinity	mg/L as $CaCO_3$	410	410	410
Chloride	mg/L	560	560	560
Sulfate	mg/L as SO_4	100	100	100
Sodium	mg/L	320	320	320
Potassium	mg/L	27	27	27
Calcium	mg/L	120	120	120
Magnesium	mg/L	52	52	52
Ammonia	mg/L as N	30	0.25	0.25
Phosphorus	mg/L	14	0.8	0.3
Iron	mg/L	Not reported	Not reported	0.1[4]
Manganese	mg/L	Not reported	Not reported	0.1[4]

[1] WRRF influent and effluent concentrations compiled by MPCA for municipal WRRFs in Minnesota.

[2] Conventional activated sludge with nitrification, secondary clarification, and no coagulation or tertiary treatment, compiled by MPCA for municipal WRRFs in Minnesota without significant industrial PFAS inputs.

[3] pH assumed to be neutral, and TDS assumed to be in typical range for WRRF effluent from regions sourcing drinking water from groundwater sources.

[4] Based on the pretreatment discussion in Section 5.2. Refer to Section 6.5 for more guidance on potential retrofit options to achieve this water quality prior to PFAS separation technologies.

6.2 PFAS Management Alternatives Evaluation Results

Each PFAS management alternative identified for liquid-phase PFAS separation and destruction in Section 4.1 was scored for each evaluation criterion described in Section 5.1.1 for municipal WRRF effluent.

6.2.1 Evaluation Scoring Results

Alternatives evaluated for municipal WRRF effluent, as described in Section 4.1, were:

- Alternative 1a: GAC sorption followed by off-site, high-temperature incineration of GAC

- Alternative 1b: RO membrane separation followed by GAC sorption of concentrate and off-site, high-temperature incineration of GAC
- Alternative 2a: single-use AIX followed by off-site, high-temperature incineration of resin
- Alternative 2b: RO membrane separation followed by single-use AIX of concentrate and off-site, high-temperature incineration of AIX resin
- Alternative 3a: regenerable AIX followed by on-site brine regeneration of resin and off-site, high-temperature incineration of spent regenerant residuals
- Alternative 3b: RO membrane separation followed by regenerable AIX sorption of concentrate and on-site brine regeneration of resin and off-site, high-temperature incineration of spent regenerant residuals
- Alternative 4a: regenerable AIX followed by on-site brine regeneration of resin and SCWO of still bottoms
- Alternative 4b: RO membrane separation followed by regenerable AIX sorption of concentrate and on-site brine regeneration of resin and SCWO of still bottoms
- Alternative 5b: RO membrane separation followed by modified clay media sorption of concentrate and off-site, high-temperature incineration of clay media
- Alternative 6a: GAC sorption followed by single-use AIX with off-site, high-temperature incineration of sorption media
- Alternative 6b: RO membrane separation with concentrate treated by GAC sorption followed by single-use AIX with off-site, high-temperature incineration of sorption media
- Alternative 7b: RO membrane separation followed by SCWO of concentrate

Evaluation results are summarized in Table 6-3 and described in subsequent sections. Specific criteria and subcriteria were defined in Section 5.1. Based on the screening, Alternatives 1a, 2a, 6a, and 6b were retained for preliminary design and cost curve development.

Table 6-3 Alternatives evaluation results for municipal WRRF effluent

Alternative Number		**1a**	**1b**	**2a**	**2b**	**3a**	**3b**	**4a**	**4b**	**5b**	**6a**	**6b**	**7b**
Membrane separation technology		-	RO	-	RO	-	RO	-	RO	RO	-	RO	RO
First PFAS sorption technology		GAC	GAC	AIX	AIX	RA	RA	RA	RA	MC	GAC	GAC	-
Second PFAS sorption technology		-	-	-	-	-	-	-	-	-	AIX	AIX	-
PFAS destruction technology		Inc	Inc	Inc	Inc	Inc	Inc	SC	SC	Inc	Inc	Inc	SC
Technical Feasibility	wt	60	55	60	55	52	49	48	45	51	66	61	*48*
PFAS separation efficiency	5	3	3	3	3	3	3	3	3	3	3	3	3
PFAS destruction efficiency	5	3	3	3	3	3	3	3	3	3	3	3	3
Degree of commercialization	4	3	3	3	3	2	2	1	1	2	3	3	1
Reliability of performance	3	1	1	1	1	3	2	3	2	1	3	3	3
Simplicity of operation/maintenance	3	3	2	3	2	1	1	1	1	2	3	2	1
Operator and public health	2	3	2	3	2	1	1	1	1	2	3	2	1
Economic Feasibility	**wt**	**44**	**39**	**44**	**39**	**33**	**26**	**21**	**21**	**30**	**37**	**32**	***16***
Relative capital costs	4	3	3	3	3	2	1	1	1	2	2	2	1
Relative operation and maintenance costs	3	3	2	3	2	3	2	2	2	2	2	1	1
Relative energy consumption	2	3	2	3	2	2	2	2	2	2	3	2	1
Relative complexity and cost of pretreatment	2	1	1	1	1	1	1	1	1	1	1	1	1
Applicability at scale	5	3	3	3	3	2	2	1	1	2	3	3	1
Byproducts Management	**wt**	**3**	**4**	**3**	**4**	**4**	**5**	**6**	**7**	**4**	**3**	**4**	***6***
Beneficial reuse opportunity for water or byproducts	1	1	2	1	2	2	3	2	3	2	1	2	2
Potential for media shifting of PFAS	2	1	1	1	1	1	1	2	2	1	1	1	2
Total Score		**107**	**98**	**107**	**98**	**89**	**80**	**75**	**73**	**85**	**106**	**97**	**70**
Retained?		**x**		**x**							**x**	**x**	

wt = criteria weight, as defined previously in Section 5.1.4, RO = RO membrane separation, GAC = granular activated carbon, AIX = anion exchange, RA = regenerable anion exchange, MC = modified clay, Inc = high-temperature incineration, SC = supercritical water oxidation (SCWO)

6.2.2 Technical Feasibility

All alternatives evaluated receive a score of 3 for **PFAS separation efficiency** because they were selected to be able to remove all target PFAS under selected operating conditions. While short-chain PFAS, such as PFBA, may break through quickly in alternatives that include media sorption technologies, treated water could potentially be kept below treatment targets given frequent monitoring and proactive changeout.

All alternatives evaluated received a score of 3 for **PFAS destruction efficiency** because both high-temperature incineration and SCWO mineralize 99% or more of target PFAS. However, both have the potential for additional PFAS formation (Krause et al. 2022; L. Winchell et al. 2022). Additional research is needed for both these technologies to demonstrate the degree of mineralization achievable and the nature of byproducts formed.

Alternatives with combinations of RO membrane separation, GAC, single-use AIX, and high-temperature incineration received a score of 3 for **degree of commercialization**, because all of these technologies have been widely applied in multiple industries for PFAS removal. AIX regeneration and modified clay sorption have been demonstrated in fewer industries and applications, so alternatives that included those technologies received a score of 2. Of the technologies included for consideration, SCWO is the least applied for PFAS management (limited to pilot-scale application for liquid treatment), so alternatives with SCWO scored a 1.

Short-chain PFAS have the potential to break through sorption media vessels between monitoring events, resulting in detectable effluent concentrations until a monitoring event and subsequent sorption media changeout can be completed. AIX resin and modified clay are more effective at removing short-chain PFAS than GAC but are also more susceptible to fouling, so all technologies using one lead-lag sorption process scored a 1 for **reliability of performance**. Regenerable AIX scored higher for reliability because on-site regeneration could provide the ability to regenerate once breakthrough is observed more quickly than single-use sorption media could be changed out. However, using regenerable AIX to treat RO concentrate scored a 2 because it is sensitive to the higher salt content in concentrate. Alternatives with both GAC and AIX scored a 3 because this multiple barrier approach provides more opportunity to capture breakthrough before it reaches final effluent. RO with SCWO of concentrate scored a 3, reflecting the high expected reliability of both these processes.

Alternatives with GAC, modified clay, or single-use AIX media and off-site high-temperature incineration scored a 3 for **simplicity of operation/maintenance** because, other than sorption media changeout events, they have only one on-site process that is relatively simple to operate. Alternatives with RO followed by sorption media all scored a 2 because RO membrane operation can be complex, requiring considerations for membrane fouling and cleaning. Regenerable AIX, with and without RO pre-concentration, scored a 1 because the regenerable AIX process includes high temperatures and chemical management. RO with SCWO of concentrate scored a 1 because of RO operational complexity and high temperature and pressure conditions associated with SCWO.

Sorption media alternatives with high-temperature incineration received a score of 3 because minimal **operator and public health** risks are associated with these types of treatment and destruction technologies. Alternatives with regenerable AIX or SCWO scored a 1 due to high-temperature on-site processes. Alternatives with RO and other media received a score of 2 because, though commonly used in the industry, they have increased risk to operators and public health due to one or more of the following: addition of chemicals, high operating temperatures, or high operating pressures.

6.2.3 Economic Feasibility

Alternatives using single-use sorption media have the lowest **relative capital cost** and received a score of 3. Adding RO as a pre-concentration step increases the capital cost for RO, but that additional cost is similar to savings resulting from smaller media filtration, leaving these scores at 3. Installation of on-site AIX regeneration equipment is expected to have a large capital cost, so alternatives with regenerable AIX technology received a lower score (by 1) than single-use AIX. Alternatives with both GAC and single-use AIX also scored 1 point lower than alternatives with just one of those technologies, resulting in a score of 2 both with or without RO. SCWO is expected to be relatively high in capital cost, so alternatives including SCWO scored a 1.

Alternatives with single-use sorption media disposed of via high-temperature incineration are expected to have the lowest **relative O&M cost (without energy)**, with primary costs associated with sorption media changeout and high-temperature incineration of media. The score for sorption media alternatives with RO upstream of the media was 1 point lower than the score without RO due to increased O&M costs associated with RO operation and maintenance. SCWO paired with regenerable AIX is expected to treat a very small waste stream consisting of AIX still bottoms, so O&M costs are ranked a 2, while RO/SCWO will send a larger flow rate to SCWO and was ranked a 1.

Alternatives with media sorption without RO concentration were the alternatives with the lowest **relative energy consumption** and had a score of 3, as energy costs would be associated with off-site high-temperature incineration of a concentrated media volume. The remaining alternatives were scored relative to these alternatives. Each alternative with RO lost a point due to the high pressures and resultant high energy needed to operate RO. Regenerable AIX would use more energy on-site for the regeneration process but less energy for residual destruction due to the low volume of residual still bottoms. These alternatives were ranked a 2. RO/SCWO ranked the lowest due to the high energy cost of applying SCWO to the full volume of RO concentrate.

All alternatives evaluated for municipal WRRF effluent were scored a 1 for **relative complexity and cost of pretreatment** because all would require pretreatment, as described in Section 6.5.

GAC, single-use AIX, and RO membrane separation have been widely **applied at scale** for municipal wastewater treatment (over 1 MGD), so alternatives including only these technologies received a score of 3. Regenerable AIX and modified clay media have limited applications at this flow rate, and alternatives including them received a score of 2. SCWO applications are very limited in size, so alternatives with SCWO were scored as 1.

6.2.4 Byproducts Management

Alternatives with two opportunities for **beneficial reuse** of byproducts received a score of 3—for example, RO permeate for water reuse in the plant and reuse of resin through regeneration in the alternative combining RO, regenerable AIX, and SCWO of still bottoms. Alternatives with one form of beneficial reuse received a score of 2. Alternatives with no potential beneficial reuse of byproducts received a score of 1.

All alternatives with high-temperature incineration received a score of 1. This is due to the current uncertainty and ongoing industry research to investigate PFAS transformations and emissions associated with high-temperature incineration. Alternatives with SCWO destruction received a score of 2 because the possibility of **PFAS media shifting** is unknown. The potential for PFAS aerosolization to occur in WRRF aeration basins should be evaluated as a phase transfer mechanism for WRRFs under all scenarios.

6.2.5 PFAS Management Alternatives for Preliminary Design

Based on the screening described in Section 6.2, four PFAS management alternatives were carried forward to preliminary design and cost analysis for municipal WRRF effluent:

- Alternative 1a: GAC sorption followed by off-site, high-temperature incineration of GAC (or GAC reactivation for WRRFs larger than 1 MGD)
- Alternative 2a: single-use AIX followed by off-site, high-temperature incineration of resin
- Alternative 6a: GAC sorption followed by single-use AIX with off-site, high-temperature incineration of sorption media (and GAC reactivation for WRRFs larger than 1 MGD)
- Alternative 6b: RO membrane separation with concentrate treated by GAC sorption followed by single-use AIX with off-site, high-temperature incineration of sorption media

While Alternatives 1b and 2b both scored 1 point higher than 6b, they were not included here, as the additive cost of RO preconcentration and cost savings on smaller media vessels will be similar for all three alternatives, and 6b reflects a moderate scenario due to its inclusion of both GAC and AIX processes. Costs for 1b and 2b could be estimated by subtracting 6a costs from 6b to get added RO costs and adding them to either 1a or 2a.

6.3 Alternative Preliminary Design and Cost Curve Development

6.3.1 Alternative 1a (Municipal Wastewater)

For Alternative 1a, pretreated WRRF effluent meeting the water quality listed in Table 6-2 flows directly to lead-lag GAC adsorption pressure vessels. For facilities with less than 80,000 pounds of GAC per changeout event, once GAC media is exhausted with respect to target PFAS (limited by PFBA and PFBS in this case), it would be sent to high-temperature incineration. Larger facilities, including the 10 MGD basis used here, are expected to use enough GAC to merit dedicated reactivation of GAC for reuse at that site only. This GAC reactivation process is expected to decrease O&M costs related to the purchase of new carbon. This process flow is summarized in Figure 6-1.

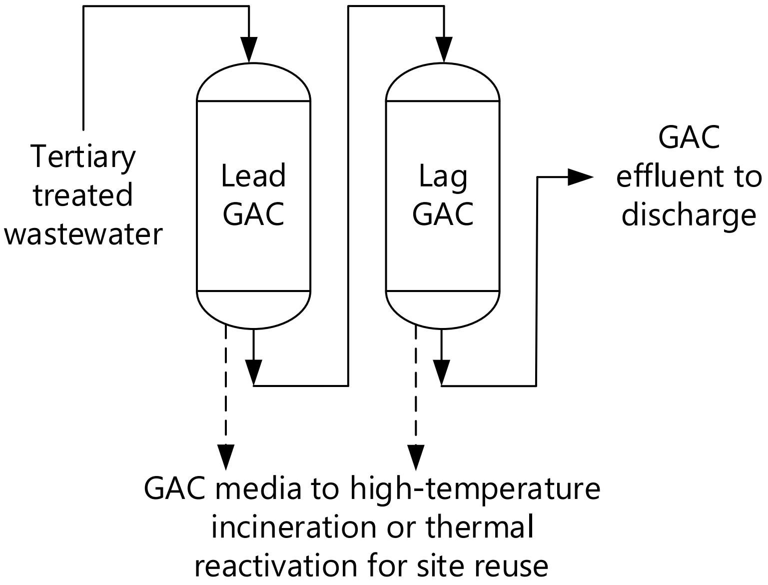


Figure 6-1 Conceptual process flow diagram for Alternative 1a for municipal wastewater

Key design basis assumptions for Alternative 1a are listed in Table 6-4, with a detailed design basis in Appendix E.

Table 6-4 Summary of design basis assumptions for Alternative 1a for municipal wastewater

Design Parameter	Basis		
	0.1 MGD/70 gpm	1 MGD/700 gpm	10 MGD/7,000 gpm
Vessel capacity (lb)	6,000	20,000	60,000
Number of trains	1	3	9
Number of vessels	2	6	18
EBCT per vessel (min)	15	15	15
HLR (gpm/sq. ft)	2.5	2.1	5.1
Estimated bed volumes to breakthrough[1]	10,000	10,000	10,000
GAC disposal route	High-temperature incineration	High-temperature incineration	Dedicated reactivation

[1] Defined as the estimated volume of water treated through the lead media vessel until the first detection of PFAS at 5 ng/L. Described further in Section 5.3.4 and Appendix D.

Figure 6-2 and Figure 6-3 illustrate estimated capital and annual O&M costs for a range of influent flow rates. These costs do not include pretreatment costs described in Section 6.5. Detailed capital and O&M cost estimates are in Appendix E.

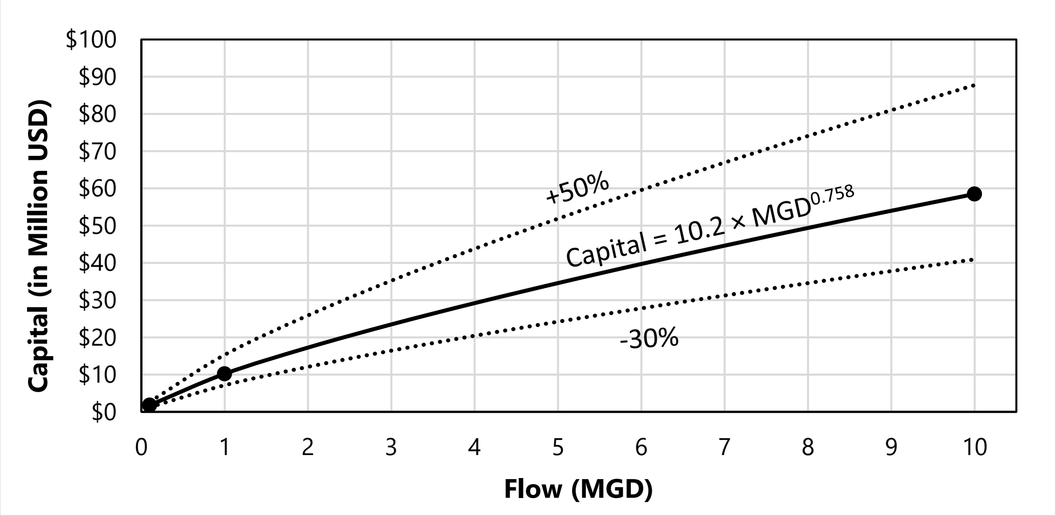


Figure 6-2 **Capital cost curve for Alternative 1a for municipal wastewater**

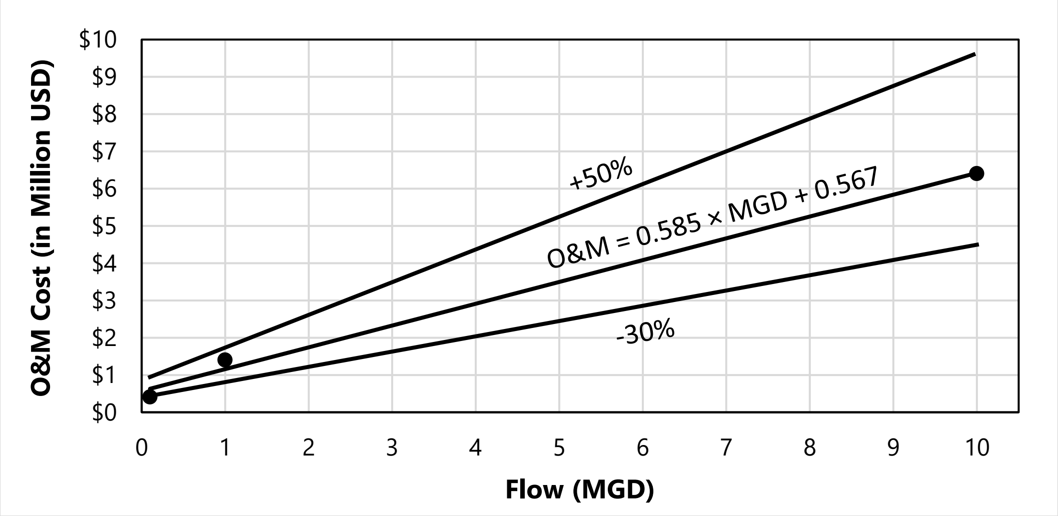


Figure 6-3 **O&M curve for Alternative 1a for municipal wastewater**

The highest operating costs are associated with labor and sorption media replacement and disposal. Primary uncertainties for Alternative 1a for municipal wastewater treatment include the following:

- Pretreatment processes needed and the ability to meet pretreatment targets
- Actual GAC breakthrough timing and requirements, depending on actual pretreated water quality
- Potential for GAC fouling limiting bed life over PFAS breakthrough
- Location and incineration fees of a selected high-temperature incineration facility

6.3.2 Alternative 2a (Municipal Wastewater)

For Alternative 2a, pretreated WRRF effluent meeting the water quality listed in Table 6-2 flows directly to lead-lag AIX pressure vessels. Once single-use AIX media is exhausted with respect to target PFAS (limited by PFBA and PFBS in this case), it would be sent to high-temperature incineration. This process flow is summarized in Figure 6-4.

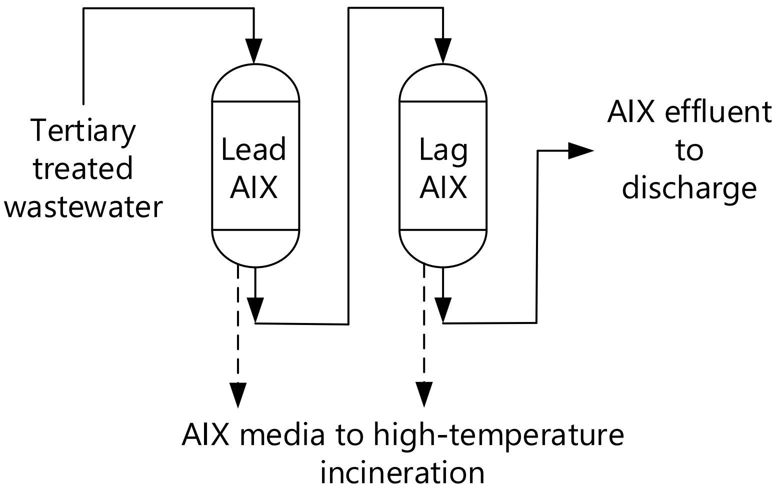


Figure 6-4 **Conceptual process flow diagram for Alternative 2a for municipal wastewater**

Key design basis assumptions for Alternative 2a are listed in Table 6-5, with a detailed design basis in Appendix E.

Table 6-5 **Summary of design basis assumptions for Alternative 2a for municipal wastewater**

Design Parameter	Basis		
	0.1 MGD/70 gpm	1 MGD/700 gpm	10 MGD/7,000 gpm
Vessel capacity (cu. ft)	40	200	450
Number of trains	2	2	9
Number of vessels	4	4	18
EBCT per vessel (min)	4	4	4
HLR (gpm/sq. ft)	5.0	7.0	6.9
Estimated bed volumes to breakthrough[1]	20,000	20,000	20,000
AIX disposal route	High-temperature incineration	High-temperature incineration	High-temperature incineration

[1] Defined as the estimated volume of water treated through the lead media vessel until the first detection of PFAS at 5 ng/L. Described further in Section 5.3.4 and Appendix D.

Figure 6-5 and Figure 6-6 illustrate estimated capital and annual O&M costs for a range of influent flow rates. These costs do not include pretreatment costs described in Section 6.5. Detailed capital and O&M cost estimates are in Appendix E.

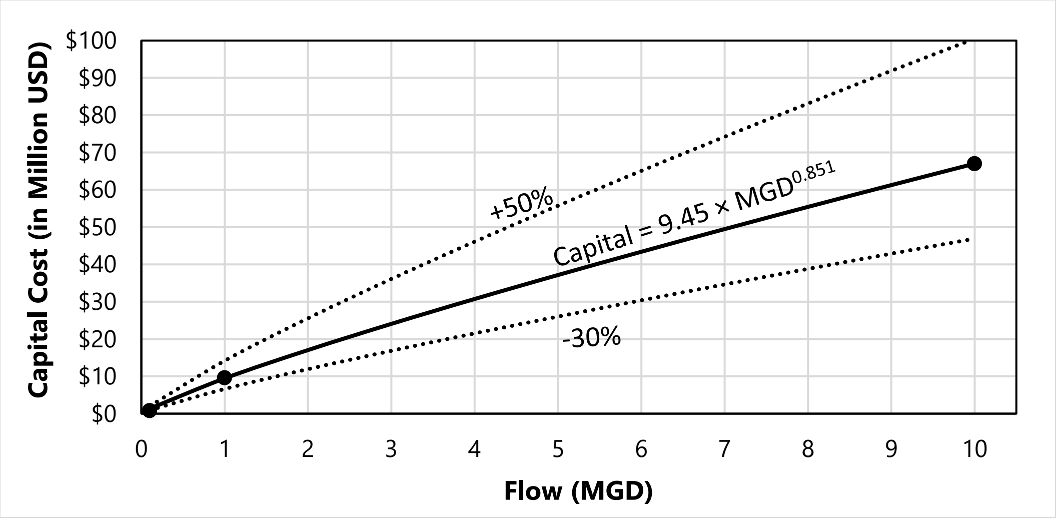


Figure 6-5 **Capital cost curve for Alternative 2a for municipal wastewater**

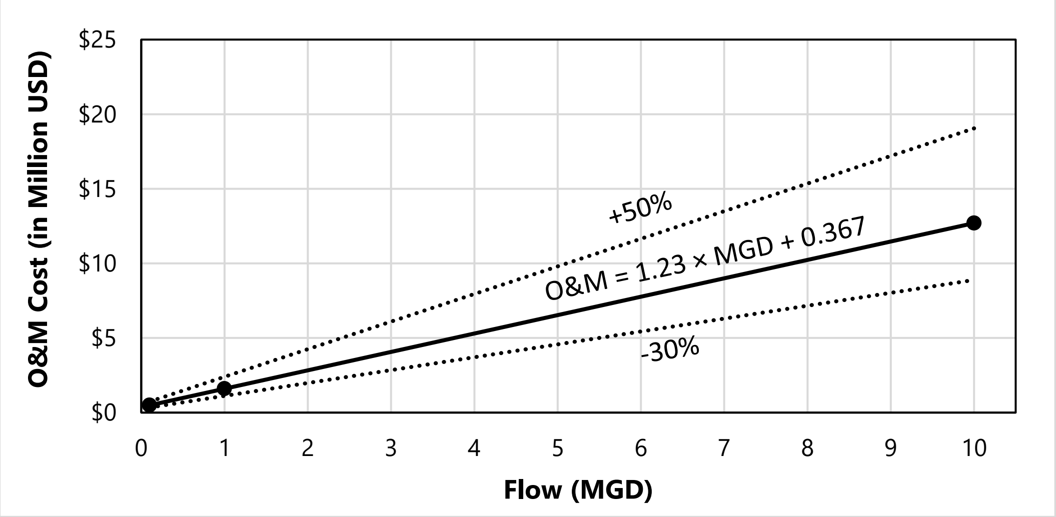


Figure 6-6 **O&M cost curve for Alternative 2a for municipal wastewater**

Estimated capital costs for single-use AIX (Alternative 2a) are similar to those estimated for GAC sorption (Alternative 1a). O&M cost estimates are similar between Alternatives 2a and 1a at low flows but diverge at higher flows because in Alternative 1a spent GAC is assumed to be reactivated rather than incinerated in the 10 MGD cost estimate, which substantially decreases the O&M costs.

Primary uncertainties for Alternative 2a for municipal wastewater treatment include the following:

- Pretreatment processes needed and the ability to meet pretreatment targets
- Actual AIX breakthrough timing and requirements, depending on actual pretreated water quality
- Potential for AIX fouling limiting bed life over PFAS breakthrough
- Location and incineration fees of the selected high-temperature incineration facility

6.3.3 Alternative 6a (Municipal Wastewater)

For Alternative 6a, pretreated WRRF effluent meeting the water quality listed in Table 6-2 flows directly to lead-lag GAC adsorption pressure vessels. GAC effluent would be routed to AIX pressure vessels. For facilities with less than 80,000 pounds of GAC per changeout, once GAC media is exhausted with respect to target PFAS (limited by PFBA and PFBS in this case), it would be sent to high-temperature incineration. Larger facilities are expected to use enough GAC to merit dedicated GAC reactivation, which enables GAC reactivation and reuse for that site only. This GAC reactivation process is expected to decrease O&M costs related to the purchase of new carbon. The single-use AIX process would primarily be present to limit the risk of breakthrough to treated effluent; GAC would be changed out frequently enough to target no breakthrough. Single-use AIX resin changeout is expected to be infrequent. Spent AIX resin would be sent to high-temperature incineration. This process flow is summarized in Figure 6-7.

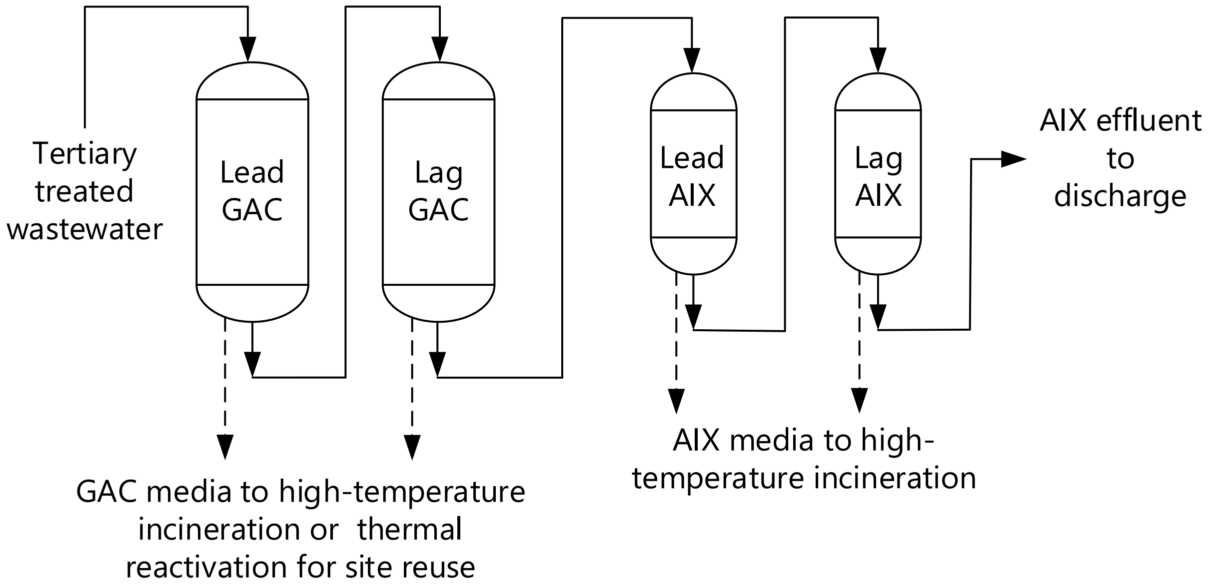


Figure 6-7 Conceptual process flow diagram for Alternative 6a for municipal wastewater

Key design basis assumptions for Alternative 6a are listed in Table 6-6, with a detailed design basis in Appendix E.

Table 6-6 Summary of design basis assumptions for Alternative 6a for municipal wastewater

Design Parameter	Basis		
	0.1 MGD/70 gpm	1 MGD/700 gpm	10 MGD/7,000 gpm
GAC			
Vessel capacity (lb)	6,000	20,000	60,000
Number of trains	1	3	9
Number of vessels	2	6	18
EBCT per vessel (min)	15	15	15
HLR (gpm/sq. ft)	2.5	4.6	5.1
Estimated bed volumes to breakthrough[1]	10,000	10,000	10,000
GAC disposal route	High-temperature incineration	High-temperature incineration	Dedicated reactivation
AIX			
Vessel capacity (cu. ft)	40	200	450
Number of trains	2	2	9
Number of vessels	4	4	18
EBCT (min)	4	4	4
HLR (gpm/sq. ft)	5.0	7.0	6.9
Estimated bed volumes to breakthrough[2]	140,000	140,000	140,000
AIX disposal route	High-temperature incineration	High-temperature incineration	High-temperature incineration

[1] Defined as the estimated volume of water treated through the lead media vessel until the first detection of PFAS at 5 ng/L. Described further in Section 5.3.4 and Appendix D.

[2] Bed volumes to breakthrough shown assume one media bed replacement per year. Described further in Appendix D.

Figure 6-8 and Figure 6-9 illustrate estimated capital and annual O&M costs for a range of influent flow rates. These costs do not include pretreatment costs described in Section 6.5. Detailed capital and O&M cost estimates are in Appendix E.

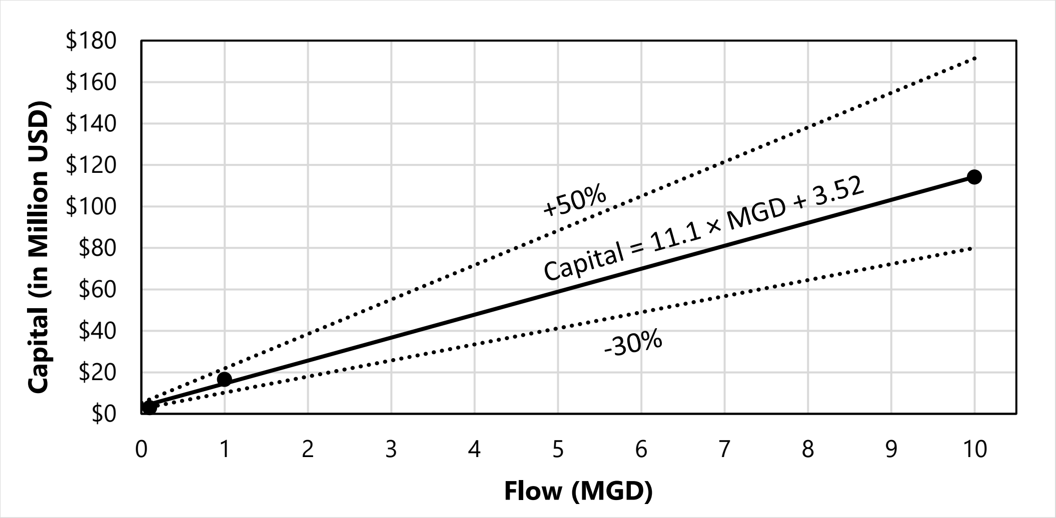


Figure 6-8 **Capital cost curve for Alternative 6a for municipal wastewater**

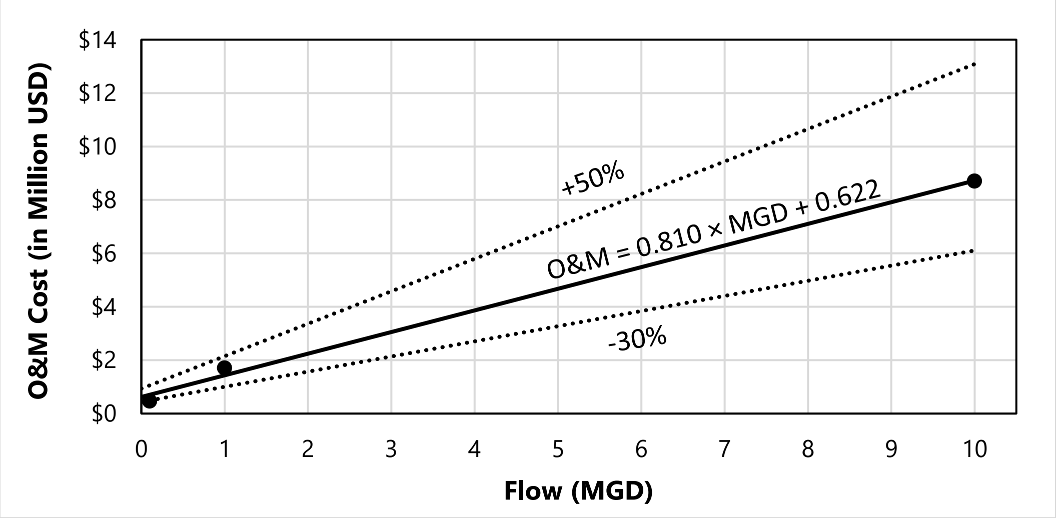


Figure 6-9 **O&M cost curve for Alternative 6a for municipal wastewater**

Estimated capital costs for the two-media sorption system, including both GAC and AIX (Alternative 6a), are approximately double those estimated for either sorption media alone (Alternatives 1a and 2a) due to the need for twice as many vessels. However, O&M costs are only slightly higher than those presented for GAC only because AIX resin changeout is expected to be infrequent.

Primary uncertainties for Alternative 6a for municipal wastewater treatment include the following:

- Actual GAC breakthrough timing and requirements, depending on actual pretreated water quality
- Actual AIX breakthrough timing and requirements, depending on actual GAC effluent water quality
- Potential for GAC fouling, limiting bed life over PFAS breakthrough
- Relative balancing of changeout priority for GAC versus AIX media

- Pretreatment processes needed and an ability to meet pretreatment targets

- Location and incineration fees of a selected high-temperature incineration facility

6.3.4 Alternative 6b (Municipal Wastewater)

Alternative 6b includes both GAC and AIX sorption processes, as previously described for Alternative 6a, with the addition of an RO membrane pre-concentration step. Tertiary treated water would route through an RO membrane separation process, with membrane concentrate routed to the lead-lag GAC adsorption pressure vessels followed by AIX pressure vessels. Membrane permeate would be blended with AIX effluent for discharge. This process flow is summarized in Figure 6-10.

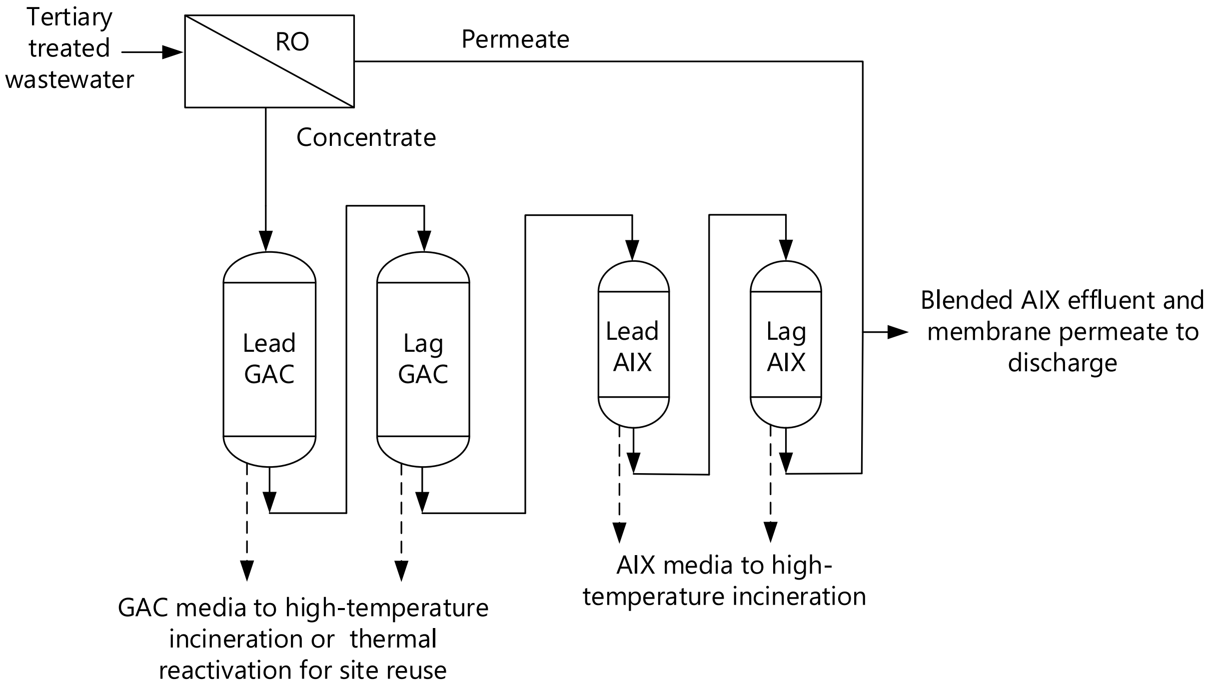


Figure 6-10 Conceptual process flow diagram for Alternative 6b for municipal wastewater

Key design basis assumptions for Alternative 6b are listed in Table 6-7, with a detailed design basis in Appendix E.

Table 6-7 Summary of design basis assumptions for Alternative 6b for municipal wastewater

Design Parameter	Basis		
	0.1 MGD/70 gpm	1 MGD/700 gpm	10 MGD/7,000 gpm
RO			
Recovery (%)	85	85	85
Flux (gal./sq. ft/day)	16	16	16
GAC			
Vessel capacity (lb)	800	6,000	20,000
Number of trains	2	2	4
Number of vessels	4	4	8
EBCT per vessel (min)	15	15	15
HLR (gpm/sq. ft)	3.2	1.9	2.4
Estimated bed volumes to breakthrough[1]	8,100	8,100	8,100
GAC disposal route	High-temperature incineration	Dedicated reactivation	Dedicated reactivation
AIX			
Vessel capacity (cu. ft)	5	40	200
Number of trains	3	2	3
Number of vessels	6	4	6
EBCT per vessel (min)	4	4	4
HLR (gpm/sq. ft)	3.8	7.8	7.3
Estimated bed volumes to breakthrough[2]	140,000	140,000	140,000
AIX disposal route	High-temperature incineration	High-temperature incineration	High-temperature incineration

[1] Defined as the estimated volume of water treated through the lead media vessel until the first detection of PFAS at 5 ng/L. Described further in Section 5.3.4 and Appendix D.

[2] Bed volumes to breakthrough shown assume one media bed replacement per year.

Figure 6-11 and Figure 6-12 illustrate estimated capital and annual O&M costs for a range of influent flow rates. These costs do not include pretreatment costs described in Section 6.5. Detailed capital and O&M cost estimates are in Appendix E.

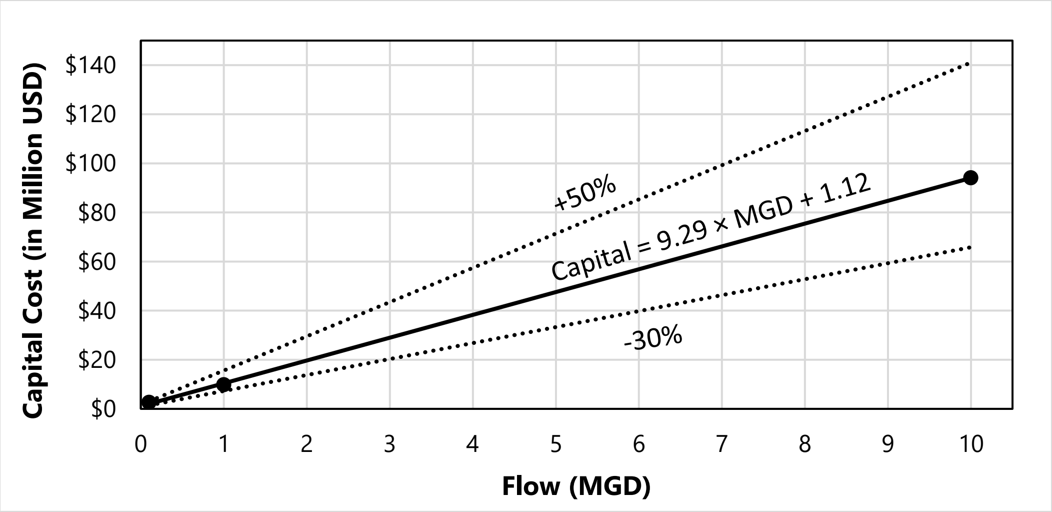


Figure 6-11 **Capital cost curve for Alternative 6b for municipal wastewater**

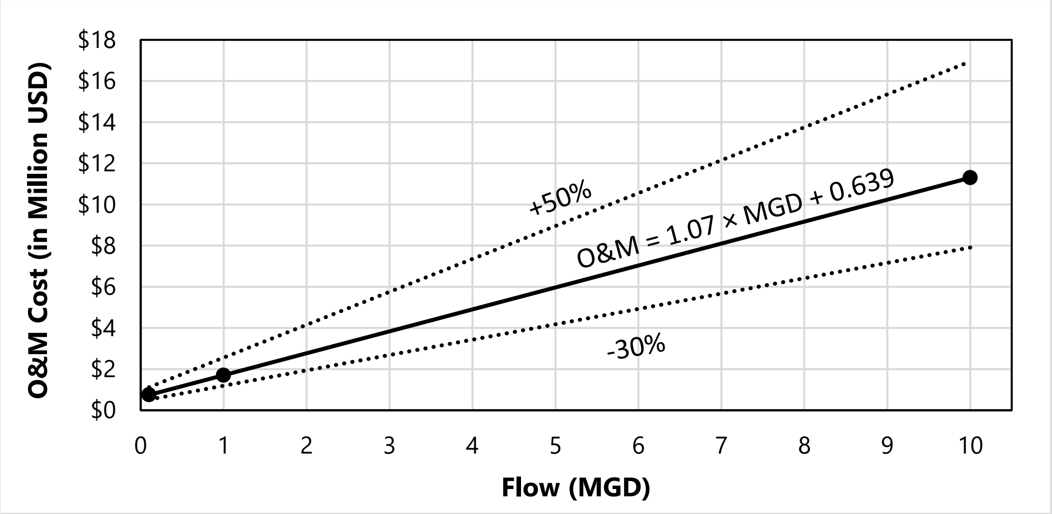


Figure 6-12 **O&M cost curve for Alternative 6b for municipal wastewater**

Estimated capital costs for the RO plus a two-media adsorption system, including both GAC and AIX (Alternative 6b), are lower than those estimates for the two-media system without RO (Alternative 6a). While the RO membrane equipment adds additional costs, the sorption media vessel sizing can be much smaller because the media influent flow is the membrane concentrate, which is lower than the total treated effluent flow routed to sorption media for Alternative 6a. O&M costs for the system with RO are higher than Alternative 6a due to similar media use rates but added O&M costs for RO operation.

Primary uncertainties for Alternative 6a for municipal wastewater treatment include the following:

- Achievable recovery of RO membranes and associated energy usage
- Fouling potential of RO membranes, depending on actual pretreated water quality

- Fouling potential of GAC and AIX media treating RO concentrate as a result of concentrated salts and metals
- Potential for GAC fouling limiting bed life over PFAS breakthrough
- Actual GAC breakthrough timing and requirements, depending on actual pretreated water quality
- Actual AIX breakthrough timing and requirements, depending on actual GAC effluent water quality
- Relative balancing of changeout priority for GAC versus AIX media
- Pretreatment processes needed and ability to meet pretreatment targets
- Location and incineration fees of selected high-temperature incineration facility

6.4 PFAS Removal Performance and Reliability

The relative ability of each alternative to meet treatment goals of <5 ng/L for target PFAS is summarized in Table 6-8. The single GAC or AIX sorption process included with Alternative 1a or 2a is expected to meet limits the majority of the time, with some risk for breakthrough of short-chain PFAS between monitoring events. Including both GAC and AIX processes in series increases the reliability of consistent removal by providing an additional treatment buffer before discharge. Including RO (Alternative 6b versus 6a) has limited effect on PFAS removal because the PFAS removal mechanism is the same GAC sorption process. Specific references for PFAS removal efficiency are detailed further in Appendix C.

Table 6-8 PFAS removal performance and reliability for municipal WRRF effluent treatment alternatives

Target PFAS	Assumed Influent Concentration (ng/L)	Percent Removal to Achieve <5 ng/L	Relative Ability to Consistently Meet Targets			
			1a: GAC	2a: AIX	6a: GAC/AIX	6b: RO with GAC/AIX
PFBA	15	67%	Moderate	Moderate	High	High
PFBS	15	67%	Moderate	Moderate	High	High
PFHxA	30	83%	Moderate	Moderate	High	High
PFHxS	5	0%	High	High	High	High
PFOA	40	88%	High	High	High	High
PFOS	5	0%	High	High	High	High
6:2 FTS	5	0%	High	High	High	High

6.5 Example Pretreatment and Retrofit Options for PFAS Separation Technologies Applied to WRRF Effluent

As described in Section 6.1.2, the effluent water from WRRFs used in this study will need to be pretreated with tertiary treatment (filtration) to achieve pretreatment targets that will limit the fouling of downstream PFAS removal processes. Due to the inclusion of RO membrane separation or sorption media pressure vessels, all PFAS management alternatives evaluated for municipal WRRF effluents are expected to require pretreatment, either in the form of a process upgrade or a new treatment process.

One example of a tertiary treatment upgrade that could meet pretreated water quality parameters is a membrane bioreactor (MBR). The MBR process is similar to a conventional activated sludge system, but active biomass is retained using microfiltration membranes rather than a separate secondary clarifier (settling tank). The MBR configuration permits the operation of the activated sludge process with a higher biomass concentration, which reduces footprint needs. A microfiltration membrane also provides high-quality effluent with lower concentrations of suspended solids and organic material than conventional activated sludge with settling. Only eight WRRFs in Minnesota have MBR processes installed, and only four of those treat their entire flows with MBR.

This section summarizes estimated costs to achieve pretreatment needs using MBR when starting with municipal wastewater treated with activated sludge or by stabilization ponds. Retrofitting a WRRF using conventional activated sludge to include tertiary filtration, potentially with coagulation and flocculation, also has the potential to meet pretreatment targets, depending on site-specific conditions. MBRs were chosen over adding a tertiary media filtration process to a conventional activated sludge facility because the costs are expected to be similar, especially when nutrient removal is also required, and MBR provides higher quality effluent with respect to solids concentrations (Brepols, Schäfer, and Engelhardt 2010; Young et al. 2014).

The conceptual pretreatment cost estimates did not include a detailed feasibility study or any design for site-specific requirements. Pretreatment needs are expected to vary by facility due to differences in existing infrastructure, water quality, space availability, and other site-specific constraints. Thus, the degree of detail included in this Report for pretreatment is less than that provided for PFAS treatment alternatives in subsequent sections. The conceptual pretreatment costs shown are intended to provide an order-of-magnitude estimate for potential pretreatment costs. **Facilities looking to implement PFAS treatment should conduct site-specific treatment evaluations prior to capital planning or implementation.**

While existing PFAS separation technologies commercially available at the scale of WRRF effluent treatment require tertiary treatment quality upstream of PFAS separation, future and developing PFAS removal technologies may have lower pretreatment requirements. These technologies may be able to treat WRRF secondary effluent without additional tertiary treatment or could potentially augment PFAS removal during secondary treatment.

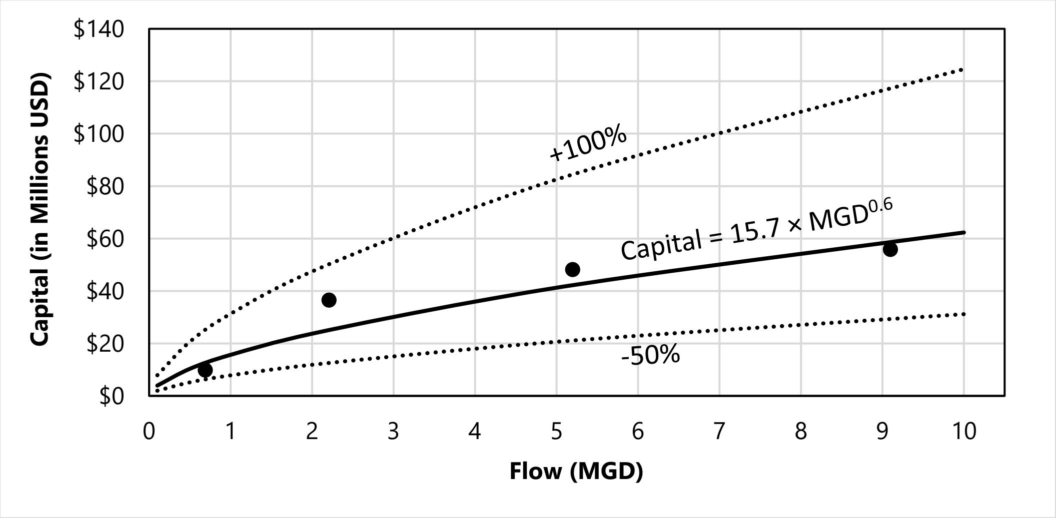
6.5.1 Example Pretreatment Retrofit for Existing Activated Sludge Treatment

Minnesota has over 300 activated sludge municipal WRRFs continuously treating wastewater. Retrofitting one of these plants with MBR technology could be done by using existing aeration basins and equipment, with membranes added directly to the existing aeration tanks, within the existing clarifier, or in separate new tanks. An MBR retrofit may also simultaneously increase treatment capacity while improving treatment performance. The approach and costs described below describe the retrofit of a typical activated sludge system to include an MBR. However, this approach will not be appropriate for all systems. The evaluation and selection of pretreatment and retrofit processes should be undertaken specifically for each facility seeking to implement a PFAS management approach.

The cost estimates developed for this Report include the following changes and additions for retrofitting an activated sludge WRRF to an MBR operation:

- Retrofit of existing aeration basins to fit membranes
- Purchase and installation of membranes
- Decommissioning or conversion of secondary clarifiers
- Installation of additional blower capacity for air scour and increased process aeration

While retrofit costs will vary significantly among WRRFs, converting an existing activated sludge facility with secondary clarification to MBR could range from 50% to 75% of the cost of a new MBR facility for a given flow capacity. Figure 6-13 presents a Class 5 (AACE) cost curve (+100%/-50%) for capital costs to retrofit a conventional activated sludge treatment facility to MBR treatment as described above. These costs are based on four MBR retrofits reported in Minnesota between 2016 and 2022 (Kyser 2022) and adjusted to November 2022 costs using the Engineering News Record (ENR) Construction Cost Index (CCI). The modeled cost curve was forced to a power factor of 0.6, consistent with the conventional 0.6 power factor for Class 5 (AACE) wastewater cost estimating (Tribe and Alpine 1986) and similar to the empirical power factor observed (0.67).



Black dots reflect actual costs for retrofit installations in Minnesota. The dashed line reflects the estimated cost curve. Black lines reflect +100%/-50% cost range. All costs scaled to November 2022 using ENR CCI.

Figure 6-13 Capital cost curve for retrofitting conventional activated sludge to MBR

Additional considerations for retrofitting a conventional activated sludge WRRF to MBR are listed below (Bagg 2009; Brepols et al. 2008):

- Flat-sheet or hollow fiber membranes—hollow fiber at lower depths
- Existing tank sizing and configuration—put membranes in or add separate membrane tank
- Upgraded screening to remove particles (from 3–12 mm to <0.5–3 mm screens) and associated headloss
- Hydraulic profile updates with membrane and screening addition and removal of clarifiers
- Energy demand of membrane air scour and increased process aeration needs for higher MLVSS concentration
- Higher RAS flow rates needed for MBRs than conventional activated sludge
- Foam management
- Nutrient removal processes and requirement
- Access for installation, maintenance, and operation

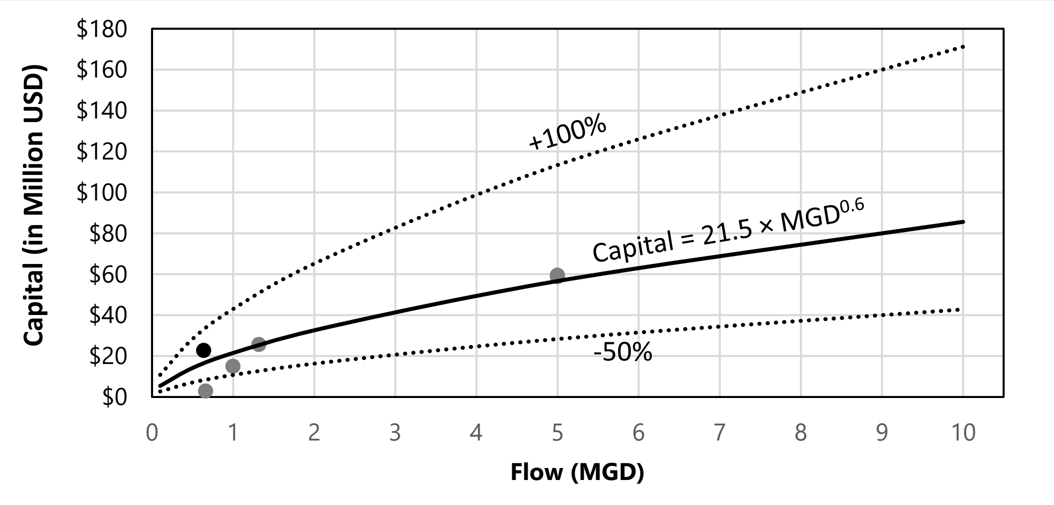
6.5.2 Example Pretreatment Retrofit for Existing Stabilization Ponds

Retrofitting an existing stabilization pond WRRF to include tertiary filtration would require significantly more investment and expansion than retrofitting an activated sludge WRRF and could be more similar to a greenfield installation than a retrofit. More WRRFs in Minnesota have stabilization ponds than activated sludge, especially in greater Minnesota, due to lower wastewater production rates and higher land availability in small communities.

The cost estimates developed in this Report assume that upgrading a stabilization pond facility includes the following changes and additions to accommodate an MBR system:

- Primary treatment, including screening, grit removal, and primary clarification (potentially could repurpose stabilization ponds)
- Aeration basins and diffusers
- Blowers and aeration control equipment
- MBR membranes
- Operational control system with an operator interface and associated measurement devices
- Pond decommissioning

For this study, converting a stabilization pond WRRF to an MBR system was assumed to require 90%–100% of the full cost of a new MBR facility for a given flow capacity. Figure 6-14 shows a Class 5 (AACE) cost curve (+100%/-50%) for capital costs to retrofit a stabilization pond WRRF to MBR treatment as described above. These estimates were based on reported MBR capital costs for one installation in Minnesota (Kyser 2022) and literature reports of MBR costs (Young et al. 2014; DeCarolis et al. 2007; Lo, McAdam, and Judd 2015; Brepols, Schäfer, and Engelhardt 2010). All were adjusted to November 2022 costs using the ENR CCI and standardized to include allowances for mechanical, electrical, piping, site work, and contractor overhead; profit, contingency, and engineering; and legal and administration fees.



Black dot reflects the actual cost for one new MBR installation in Minnesota. Grey dots reflect costs from Young et al. 2014; DeCarolis et al. 2007; Lo, McAdam, and Judd 2015; Brepols, Schäfer, and Engelhardt 2010. The solid line reflects the estimated cost curve. Dashed lines reflect +100%/-50% cost range. All costs scaled to November 2022 using ENR CCI.

Figure 6-14 Capital cost curve for new MBR installation for a stabilization pond system, including the cost to replace stabilization ponds

Other potential considerations for retrofit of a stabilization pond WRRF to an MBR system would include the activated sludge retrofit considerations listed in Section 6.5.1 as well as the following:

- Depending on the preferences of the WRRF, the existing stabilization ponds or a smaller pond footprint could be used for primary treatment and/or equalization ahead of the MBR instead of the mechanical primary treatment included above.

6.6 Other Considerations for Municipal WRRF Effluent

Which PFAS management alternative is most economically feasible for specific WRRFs will depend upon site-specific capacity, treatment goals, space availability, and operational preferences.

- Activated sludge WRRFs with flows higher than 1 MGD are likely to see the economic benefit of including an RO concentration step upstream of media sorption or AIX, as proposed in Alternative 6b. The payback period for RO will be less favorable for smaller facilities.
- Specific treatment targets will also affect technology selection, with facilities targeting primarily long-chain compounds more able to rely upon one media sorption process. Facilities targeting short-chain compounds at low concentrations or with strict permit limits disallowing daily exceedances may benefit from multiple media sorption processes in series, as proposed in Alternatives 6a and 6b.
- The decision to select one of these PFAS management alternatives and implement it at a WRRF will vary site by site based on the PFAS management strategy, existing infrastructure, existing effluent water quality, and actual treatment targets. Another consideration is that municipal WRRF effluent has a larger volume and lower PFAS concentrations than other liquid municipal waste streams evaluated in this study. The Metro Plant WRRF in Saint Paul, Minnesota, produces over 300 MGD of treated effluent. This relatively large flow with dilute PFAS concentrations highlights the challenges of downstream PFAS management. Future evaluation of PFAS management in WRRF effluent should consider further concentration processes, especially those with limited pretreatment needs. For example, foam fractionation presents a potentially appealing alternative because the volume of concentrated PFAS waste produced could be as low as a millionth of the initial water volume. However, additional studies to improve short-chain PFAS removal and equipment scale-up concerns limit current applications of foam fractionation for treating WRRF effluent.
- Large WRRFs implementing PFAS removal from effluent may be justified in building on-site destruction facilities for sorption media, foam fractionation foamate, or RO concentrate. Smaller WRRFs will need to use either existing or future regional facilities developed for that purpose.

7 Municipal Wastewater Biosolids

Management of municipal WRRF biosolids in Minnesota is currently accomplished using a combination of land application, landfilling, reed bed disposal, and sewage sludge incineration (SSI) (Beecher et al. 2022). While SSI has demonstrated the capability to destroy PFAS in some studies, depending on the design and operating conditions (L. J. Winchell et al. 2021b), the SSIs in Minnesota do not operate at the temperatures required for PFAS destruction. To date, none of the biosolids management techniques practiced in Minnesota destroy PFAS. Technologies that could be added to existing WRRFs to destroy PFAS in solids are discussed in this section.

7.1 Project Assumptions for Municipal Wastewater Biosolids

7.1.1 Description and Scale of Required Treatment

Biosolids are the byproducts of wastewater treatment (including microorganisms grown within the WRRF and solid materials) that have been stabilized in an additional process, such as anaerobic digestion. Wastewater solids treatment techniques vary depending on WRRF treatment processes and the solids end-use plan. The majority of WRRFs in Minnesota use one of the following wastewater treatment processes:

- Conventional activated sludge treatment, including grit removal, aeration, sludge settling, disinfection, and solids stabilization via anaerobic digestion. Some WRRFs include primary clarification, which produces primary sludge. Conventional activated sludge treatment results in the continuous production of solids from wasted activated sludge.

- Stabilization pond treatment, including treatment in unaerated stabilization ponds, disinfection, and in-situ anaerobic digestion of solids that have settled to the bottom of the ponds. Solids in stabilization ponds digest naturally, decreasing in volume over time. Stabilization pond treatment systems are typically designed to accumulate solids for up to five years. When the solids blanket in the pond reaches the cleanout level, biosolids are either pumped from the pond or dredged from the dewatered pond using front-end loaders.

Both treatment processes typically result in biosolids with 3–5% solids (by volume). Biosolids treatment for PFAS removal considered in this study was assumed to begin at the point of biosolids discharge from the stabilization process (i.e., anaerobic digester) at a conventional activated sludge WRRF or the stabilization pond.

Biosolids production is dependent on each WRRF’s unique influent loading and operating conditions. Each WRRF has a unique influent solids flow rate, activated sludge production rate, and volatile solids reduction rate. A WRRF with an influent flow of 1 MGD could produce approximately 0.08 to 1.0 dry tons of wastewater solids per day (dtpd) based on Minnesota-specific data. A general rule of thumb used in the remainder of this Report is to assume that a 1 MGD WRRF produces approximately 1 dtpd of biosolids.

7.1.2 Biosolids PFAS Concentrations and Treatment Targets

Target PFAS selection by waste stream was detailed previously in Section 2.3. The PFAS identified as most relevant in biosolids include long-chain PFAAs and PFAS precursors that partition to biosolids. PFAS presence within a WRRF and associated biosolids are variable and reflective of ongoing consumer and industry changes (EGLE 2020). Typical and high concentrations of PFAS in biosolids used to support the design basis and cost estimates for this study are summarized in Table 7-1. These values are not intended to reflect the full range of PFAS concentrations that can be present in WRRF biosolids. The treatment target for the selected PFAS compounds is 5 ng/g, as described in Section 2.4, which is the detection limit for PFAS in biosolids at the time of this study.

Table 7-1 Assumed influent concentrations and treatment goals for target PFAS for municipal WRRF biosolids (all units in ng/g)

Value Type	Typical [1]	High [1]	Treatment Target
PFBA	2	5	5
PFBS	20	40	5
PFHxA	5	15	5
PFHxS	15	30	5
PFOA	60	200	5
PFOS	400	600	5
PFOSA	20	70	5
N-EtFOSAA	30	60	5
N-MeFOSAA	100	200	5

[1] Data from Venkatesan and Halden (2013) and Helmer et al. (2022).

This study does not include specific treatment targets for the non-PFAS compounds in biosolids. However, the treatment methods for PFAS in biosolids will decrease both the volatile solids and water content in the biosolids.

7.1.3 Biosolids Composition and Characteristics

Table 7-2 summarizes the assumed characteristics of biosolids entering the PFAS management strategy. These characteristics were selected to represent typical municipal WRRF biosolids after anaerobic digestion. Example technologies and costs to achieve targeted solids content for pyrolysis or gasification and SCWO are discussed in Section 7.4. Biosolids have a higher concentration of organic material and nutrients than wastewater due to the higher solids content. The biosolids production rate was selected with the following considerations:

- This study's municipal WRRF influent rate was 0.1 to 10 MGD, corresponding to roughly 0.1 to 10 dtpd. Actual biosolids production rates by facility may vary significantly from this range based on processes used for wastewater treatment and solids management as well as incoming wastewater characteristics.

- The minimum solids feed rate for PFAS destruction technologies is 1 dtpd, so the range evaluated for on-site biosolids management was 1 to 10 dtpd.
- WRRFs producing less than 1 dtpd of solids could partner with other local WRRFs to accumulate enough biosolids to meet the minimum PFAS destruction process feed rate.

Table 7-2 Assumed composition and characteristics of municipal WRRF biosolids feed to PFAS destruction technologies

Parameter	Unit	Characteristics of WRRF Biosolids
Biosolids to dewatering process following anaerobic digestion or lagoon treatment process	dtpd	1–10
Total solids, pre-dewatering	%	2–4
pH	Standard Units	7–8
Total solids, dewatered for drying [1]	%	25
Total solids, dried for pyrolysis or gasification	%	90
Total solids, dewatered for SCWO	%	15

[1] Pyrolysis/gasification vendors supply the solids dryer(s) and pyrolysis/gasification unit(s) as a system package.

7.2 PFAS Management Alternatives Evaluation Results

Each PFAS management alternative identified for biosolids PFAS separation and destruction in Section 4.2 was scored for each evaluation criterion described in Section 5.1.1 for municipal WRRF biosolids. The highest individual and weighted scores indicate the most favorable alternative.

Two evaluation criteria were different for biosolids treatment than for liquid-phase treatment. PFAS separation efficiency does not apply to biosolids treatment technologies. All treatment technologies evaluated for biosolids destroy the compounds rather than separating the compounds from the biosolids. In addition, pyrolysis, gasification, and SCWO systems have opportunities to recover heat energy from the process to reduce the energy input requirements or generate electricity, so energy recovery was included as a sub-criterion.

7.2.1 Evaluation Scoring Results

Evaluation results are summarized in Table 7-3 and described in subsequent sections. Specific criteria and sub-criteria were previously defined in Section 5.1. Based on the screening, pyrolysis/gasification with thermal oxidation and SCWO were retained for preliminary design and cost curve development. The rationale for considering pyrolysis and gasification as a single technology is described in Section 7.2.5.

Table 7-3 Alternatives evaluation results for municipal WRRF biosolids

Alternative Number		**1**	**2**	**3**
PFAS separation technology		N/A	N/A	N/A
PFAS destruction technology		Pyrolysis / Thermal oxidation	Gasification / Thermal oxidation	SCWO
Technical Feasibility	**weight**	**41**	**41**	**35**
PFAS separation efficiency	N/A	N/A	N/A	N/A
PFAS destruction efficiency	5	3	3	3
Degree of commercialization	4	3	3	1
Reliability of performance	4	2	2	2
Simplicity of operation/maintenance	2	1	1	2
Operator and public health	2	2	2	2
Economic Feasibility	**weight**	**32**	**32**	**40**
Relative capital costs	4	1	1	3
Relative operation and maintenance costs	3	2	2	3
Relative energy consumption	2	2	2	3
Relative complexity and cost of pretreatment	1	2	2	2
Energy recovery options	2	3	3	3
Applicability at scale	5	2	2	1
Byproduct Management	**weight**	**13**	**13**	**13**
Beneficial reuse opportunity for water or byproducts	3	3	3	3
Potential for media shifting of PFAS	2	2	2	2
	Total Score	**86**	**86**	**88**
	Retained?	**x**[1]	**x**[1]	**x**[1]

[1] Pyrolysis and gasification with thermal oxidation are similar processes and use the same equipment. The project team determined that they could be considered together as a single technology. Additional details follow in this section.

7.2.2 Technical Feasibility

While pyrolysis, gasification, and SCWO are established technologies for treating other waste materials, using these technologies to destroy PFAS in biosolids is relatively new. Many studies are underway to improve our understanding of how to apply these technologies for PFAS in biosolids and the fate and transport of PFAS during treatment. This study evaluated the technical feasibility of treating biosolids for

PFAS using currently available data. As more research is completed, it is anticipated that future scores may be different.

All three technologies received high scores for **PFAS destruction efficiency**, with over 99% reduction in concentrations of measured PFAS compounds in treated biosolids (Thoma et al. 2022; Davis 2020; Krause et al. 2022).

Pyrolysis and gasification systems are **commercially available** and currently operating to treat municipal WRRF biosolids. They were assigned an individual score of 3. SCWO was not commercially available for biosolids treatment at the time of this study and was scored lower than the other two technologies. However, demonstration testing of a commercial prototype is expected to begin at Orange County Sanitation District in California in 2023.

All three technologies scored a 2 for **reliability of performance**. They work well within their design conditions, but variability in influent biosolids, flow rate, organics concentration, or moisture content may impact performance. In addition, the high capital cost of these systems makes it unlikely that utilities will have a duplicate standby system. If a critical component of a PFAS destruction system is inoperable, the entire process will need to stop and biosolids would need to be stored.

Pyrolysis and gasification scored lowest for **simplicity of operation/maintenance**. The technologies include thermal drying and pyrolysis/gasification, which require operator training and attention at a higher level than for a typical biosolids treatment process. SCWO requires less specialized training and was scored as a 2.

All three technologies scored a 2 for **operator and public health**, indicating they have a moderate additional health risk beyond a typical biosolids treatment process. The systems include very hot unit processes, posing increased risk to operators. In addition, thermal drying can produce dust and create an explosion hazard. SCWO systems operate at high temperatures and pressure, increasing the risk to plant staff.

The complete, weighted scoring for the three technologies resulted in pyrolysis and gasification having the same technical feasibility score of 41, while SCWO scored 35. The primary reason for the low SCWO score was its low score for commercial availability, which is highly weighted to reflect the importance of currently feasible technologies.

7.2.3 Economic Feasibility

In the categories of **relative capital cost, relative O&M costs, and relative energy consumption**, SCWO scored higher than pyrolysis and gasification. Recent vendor cost estimates per wet ton for SCWO are lower than for the other technologies. Operation and maintenance costs are also expected to be lower for SCWO due to lower energy requirements.

Pyrolysis, gasification, and SCWO all scored 2 for **relative complexity and cost of pretreatment**. All require dewatering as pretreatment prior to PFAS destruction, which is a common biosolids process but requires added equipment, operator time, and maintenance attention.

All three technologies scored high for **energy recovery**. The systems are designed to transfer heat from the treatment process exhaust gases to the initial portion of the treatment process, reducing the demand for external energy.

Currently, none of the technologies are commonly used or **available at scale** to treat PFAS in biosolids. However, pyrolysis and gasification systems are used at a few facilities, with several vendors supplying full-scale systems. Pyrolysis and gasification scored a 2 for this category to reflect a minimal commercial application of these technologies for PFAS treatment in biosolids. SCWO has no commercially available systems at this time and scored lowest in the category.

Despite the low score for SCWO's applicability at scale, the combined weighted score for economic feasibility was higher than pyrolysis or gasification, reflecting the relative importance of capital and O&M costs in the scoring for this category.

7.2.4 Byproducts Management

All three technologies scored a 3 for the **beneficial reuse of water or byproducts**. Pyrolysis and gasification produce biochar, which has multiple uses and returns heat to the process. The temperature of the operation dictates the amount of biochar produced and the energy recovered. SCWO produces clean water, inert gases, and mineral salts.

All three technologies received a score of 2 for **potential media shifting of PFAS**. The possibility of PFAS media shifting is unknown and an area of ongoing research.

7.2.5 PFAS Management Alternatives for Preliminary Design

SCWO had the highest combined weighted score (88) of the PFAS destruction technologies for biosolids. SCWO was carried forward to preliminary design and cost analysis for municipal WRRF biosolids.

The combined weighted scores for pyrolysis with thermal oxidation and gasification with thermal oxidation were the same: 86. Pyrolysis and gasification were considered similar enough in equipment, solids and energy inputs, and biochar production to be evaluated as a single alternative for preliminary design. Based on the screening, pyrolysis/gasification with thermal oxidation and SCWO were retained for preliminary design and cost curve development.

7.3 Alternative Preliminary Design and Cost Curve Development

7.3.1 Supercritical Water Oxidation (Wastewater Biosolids)

In this treatment alternative, dewatered biosolids at 15% of total solids are conveyed to the SCWO process inlet. The first component is a heat exchanger used to raise the temperature of the biosolids before SCWO. SWCO reactor effluent is routed through an economizer and liquid/gas separator.

Separated gas is used to produce energy and is vented to the atmosphere, with condensate available for discharge or reuse. Solids leaving the liquid/gas separator enter a cooler, and recovered heat is used to preheat incoming biosolids. Water is separated from resulting solids in a clarifier, with cooled solids routed to disposal or beneficial reuse. This process flow is summarized in Figure 7-1.

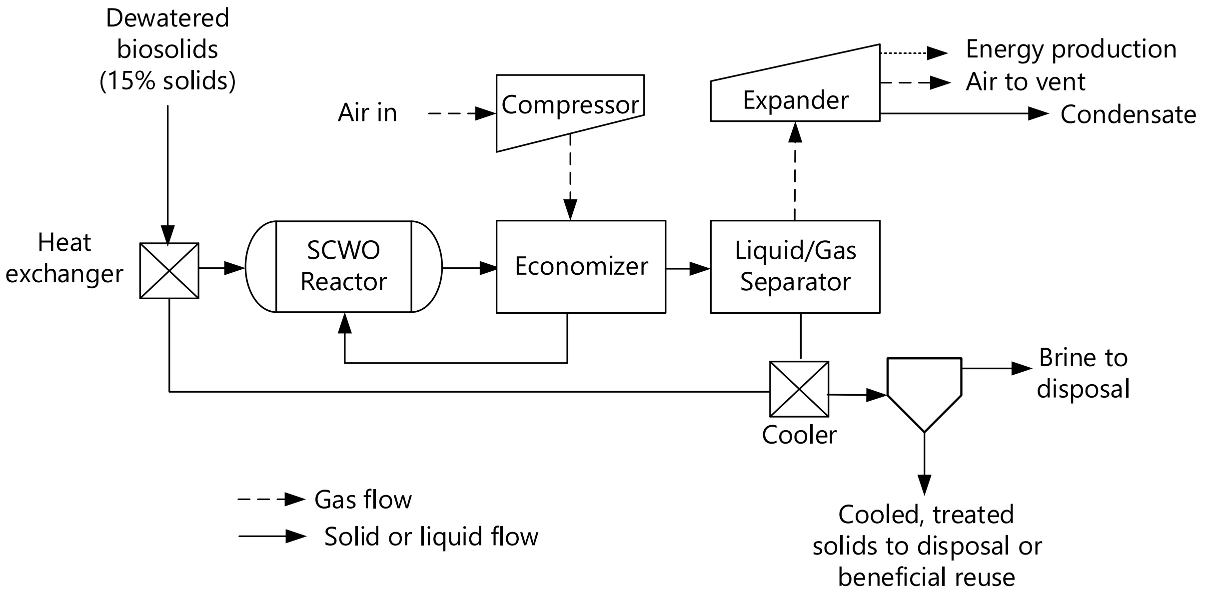


Figure 7-1 **Conceptual process flow diagram for biosolids supercritical water oxidation**

Currently, a single manufacturer is marketing SCWO systems for destroying PFAS in biosolids in the United States. The systems are sold as an equipment package and include all the unit processes shown in Figure 7-1. The manufacturer sizes the unit processes based on the rated treatment system capacity. The system sizes currently available are shown in Table 7-4.

Table 7-4 **SCWO system sizing**

Available SCWO System Sizes	Treatment Capacity, Wet Metric Tons per Day (15% total solids)	Treatment Capacity (dtpd)
SCWO 6	6	1.0
SCWO 30	30	5.0
SCWO 200	200	33

The selection of the SCWO system size for a given facility is based on the dewatered biosolids feed rate, with proposed pricing and sizing for systems between six and 200 wet tons per day (1 to 33 dtpd). The system is a continuous treatment process. External fuel is needed to bring the treatment process to operating temperature. Once at temperature, SCWO systems require minimal external fuel. Thus, running periodic treatment campaigns is not a recommended operating strategy. Facilities treating less than 1 dtpd may not be good candidates for an individual SCWO system. SCWO units can be operated in parallel to treat flow rates of biosolids that fall between the individual treatment unit capacities.

The SCWO system is provided within containerized modular units (smaller units) or on a series of skids (large unit). A contractor will connect the biosolids feed piping, electrical and controls wiring, natural gas, and water to the package treatment units. In cold-weather areas like Minnesota, the SCWO system will be enclosed within a climate-controlled building. In temperate climates, the system may be installed in a covered area but does not require a full enclosure.

The development of the cost curve for SCWO was based on the selection of the size and number of units to meet an incremental solids load increase. The SCWO system design basis and assumptions are listed in Table 7-5 and reflect manufacturer recommendations for feedstock total solids and minimum heating value.

Table 7-5 **Summary of design basis assumptions for supercritical water oxidation for biosolids**

Design Parameter	Low Loading	High Loading
Dewatered biosolids load (dtpd)	0.99–4.0	5.0–10
Total solids (%TS)	15%	15%
Minimum heating value (BTU/dry lb)	5,000	5,000
Treatment capacity (dry ton/treatment unit)	0.99	5.0
Number of treatment units	1–4	1–6
Operating schedule	24 hours, 7 days/week	
PFAS removal estimate for target PFAS	99% (Krause et al. 2022)	

Figure 7-2 illustrates estimated capital costs for a range of solids loading rates. These costs do not include pretreatment costs described in Section 7.5. Detailed capital cost estimates are in Appendix E.

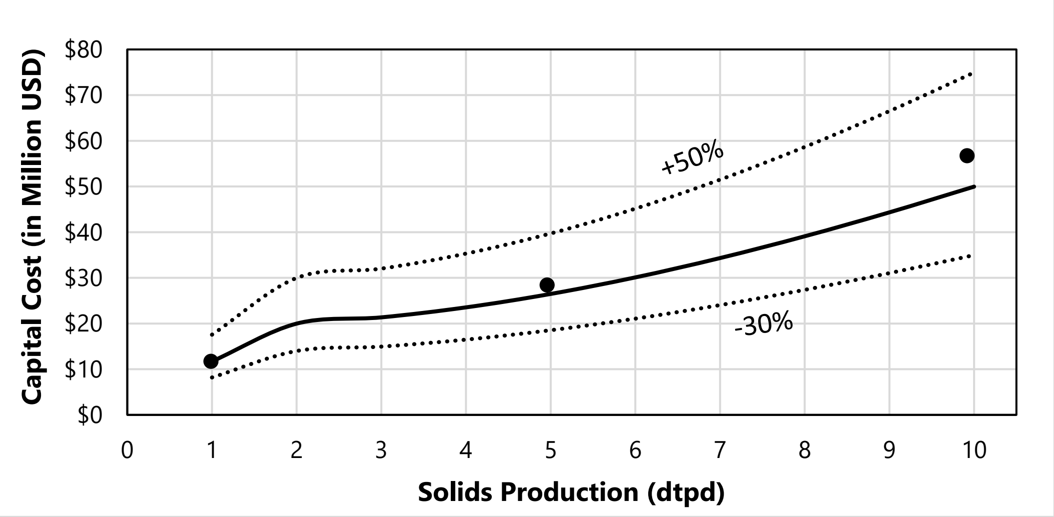


Figure 7-2 **Capital cost curve for biosolids SCWO**

The demonstration project for full-scale SCWO systems for biosolids is currently in construction, so minimal information is available on operation and maintenance costs for the full-scale operations. The SCWO manufacturer provided the predicted energy use for each unit size. The smallest SCWO systems are expected to consume 300 kWh of electricity per day. Larger SCWO systems that process over five dtpd or more are expected to produce electricity if paired with a generator. The annual O&M costs shown in Figure 7-3 include the electricity costs and savings, natural gas for startup, maintenance, materials, and labor.

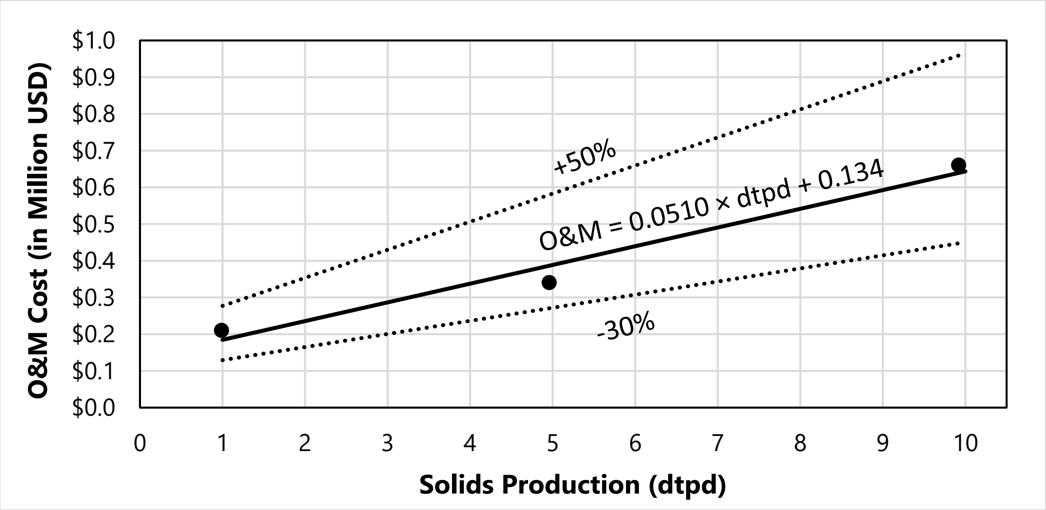


Figure 7-3 O&M cost curve for biosolids SCWO

Factors that influence the O&M costs include the heating value of the biosolids, oxidation potential, and savings from eliminating biosolids disposal costs. SCWO systems that run continuously offer energy savings by avoiding natural gas use for repeated startups; some have the potential to be energy-positive and electricity-producing.

Primary uncertainties for SCWO treatment of biosolids include the following:

- Documented commercial prototype performance with processing biosolids
- Reliability of continuous operation
- Technology robustness at a commercial scale
- Required maintenance and frequency of parts replacement
- Robustness against corrosion, plugging, and deposition of salts in the system components
- Performance of PFAS destruction at commercial scale
- Limited data available for comparing costs with conventional technologies

7.3.2 Pyrolysis/Gasification with Thermal Oxidation (Wastewater Biosolids)

As noted previously, gasification and pyrolysis are similar processes. They use similar equipment and process strategies, resulting in similar process outputs, and the costs for construction and operation are similar. Thus, both processes were evaluated as a single alternative for this study.

Pyrolysis or gasification of biosolids is a two-step process. First, dewatered biosolids at roughly 25% solids are dried to 75–90% solids in a biodryer or thermal dryer. The dried biosolids are then processed in the pyrolysis or gasification reactor unit. Literature indicates that gasification and pyrolysis reactors are operated at different temperatures. However, in practice, they are both operated at a wide range of temperatures that overlap. Gasification uses limited oxygen during operation, while pyrolysis uses no oxygen. Biochar from the pyrolyzer/gasifier is collected for beneficial use or disposal at a landfill and is documented to have non-detect PFAS. Biochar could potentially be used as a sorbent for PFAS in other waste streams, though full-scale application is limited. as discussed in more detail in Appendix B.

Pyrogas or syngas is generated from the process and contains PFAS compounds. If the gas is to be used for electricity production, it is routed to an oil/tar condenser to remove oil, tar, and other impurities. Oil and tar are routed directly to the thermal oxidizer, while the gas continues on to a scrubber before being used as generator fuel. Exhaust from the generator is returned to the thermal oxidizer and processed through the exhaust gas scrubber. If the gas is to be thermally oxidized for recoverable heat generation, oil/tar condensing and gas scrubbing upstream of the thermal oxidizer are unnecessary. Research to date has shown that PFAS are destroyed when the thermal oxidation temperature is above 1,000°C. Thermal oxidizer exhaust is processed through an additional exhaust scrubber before venting to the atmosphere. This process flow is summarized in Figure 7-4.

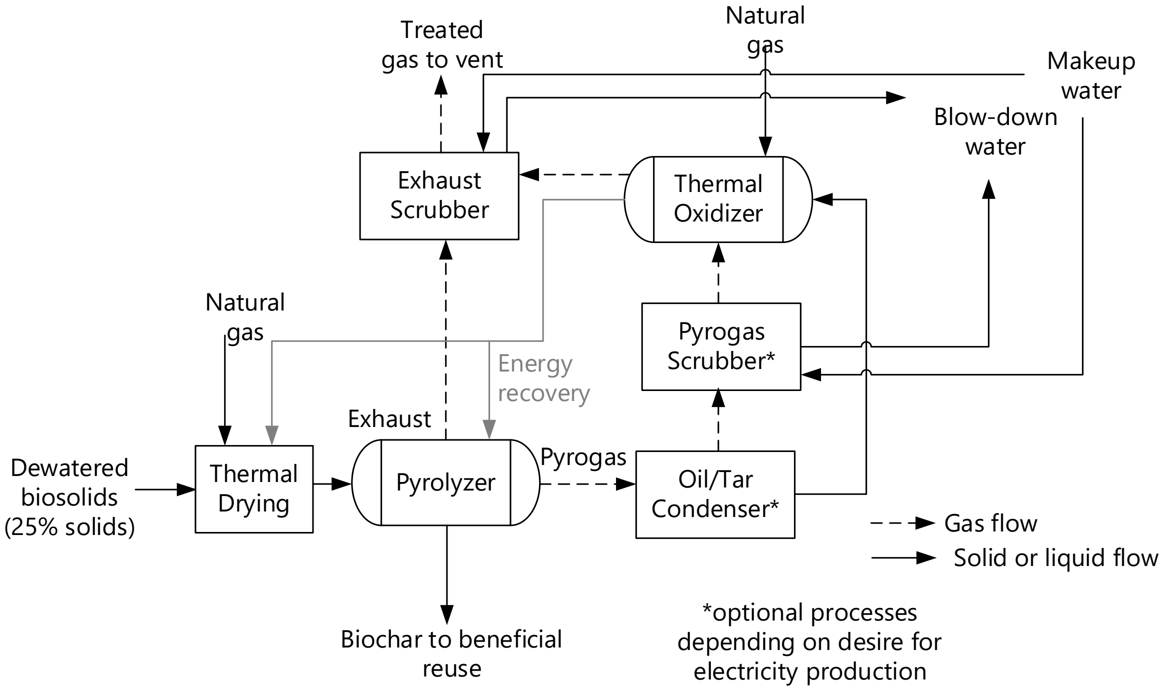


Figure 7-4 **Conceptual process flow diagram for biosolids pyrolysis/gasification with thermal oxidation**

The dryer and pyrolysis/gasification system can be provided as a complete equipment package by a single vendor. Alternatively, the pyrolysis/gasification unit can be supplied separately from the dryer if the WRRF already has a biosolids dryer. For this study, WRRFs were assumed to not have an existing biosolids dryer.

Selection of the pyrolysis/gasification system size is based on the dewatered biosolids feed rate. Continuous dryer and pyrolysis/gasification unit operation is recommended to minimize the demand for natural gas during the treatment startup process. Once the process reaches operating temperature, the energy in the biosolids may be sufficient to fuel the process without natural gas input. The minimum biosolids feed rate for autogenous (self-fueling) pyrolysis/gasification system operation is approximately 1 dtpd.

The dryer and pyrolysis/gasification units are typically separate unit processes connected by process piping. The vendor provides the connections for smaller systems. The system owner typically must provide the connections for the larger systems and the connection to the thermal oxidizer system. For all system sizes, a contractor will connect the biosolids feed piping, electrical and controls wiring, natural gas, and water to the treatment units. In cold-weather areas, the dryer and pyrolysis/gasification system would be enclosed within a climate-controlled building. In temperate climates, the system may be installed in a covered area but does not require full enclosure.

Development of the cost curve for biosolids pyrolysis/gasification systems was based on three recent pyrolysis/gasification system designs prepared for WRRFs producing 10 dtpd of biosolids or less. The primary design parameters are summarized in Table 7-6.

Table 7-6 Summary of design basis assumptions pyrolysis/gasification with thermal oxidation for biosolids

Design Parameter	Low	Mid	High
Dewatered biosolids load (dtpd)	2	6	10
Total solids (%TS)	25%	25%	25%
Evaporation rate (lb water/hour)	560	1,400	2,400
Number of dryers	1	1	1
Type of thermal dryer	Paddle	Paddle	Belt
Total solids of dried product (%TS)	90%	90%	90%
Pyrolysis/gasification unit capacity (dtpd each)	3	3	10
Number of pyrolysis/gasification units	1	2	1
Operating schedule	24 hours, 7 days/week		
Removal estimate for target PFAS	99.5%		

The resulting construction costs were plotted against the associated biosolids feed rate in dry tons per day, resulting in the cost curve shown in Figure 7-5. These costs do not include the pretreatment costs described in Section 7.5. Detailed cost tables are in Appendix E.

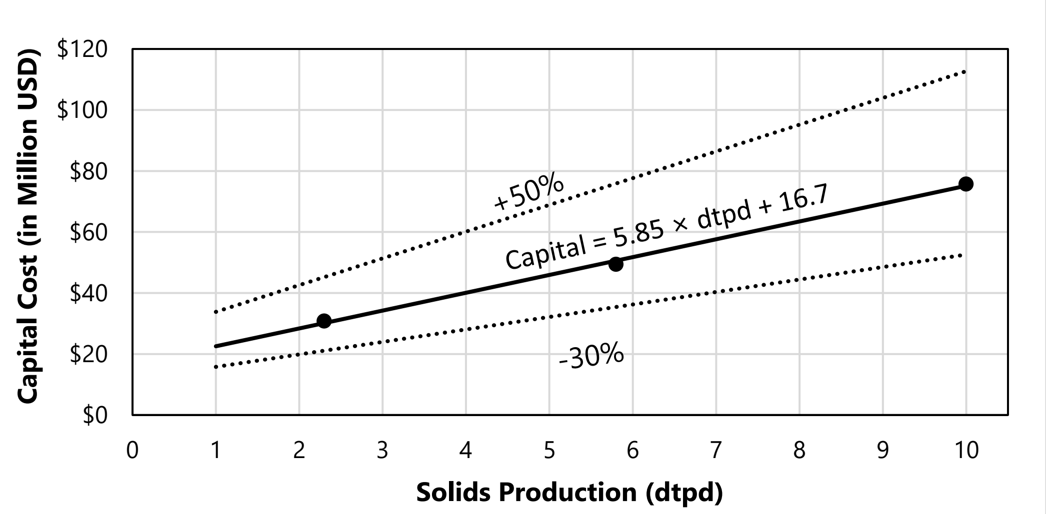


Figure 7-5 Capital cost curve for biosolids pyrolysis/gasification with thermal oxidation

Operation and maintenance costs for dryer and pyrolysis/gasification systems will vary depending on the type of dryer and pyrolysis/gasification unit selected. Recent O&M estimates prepared for pyrolysis/gasification system design projects were studied to develop a range of costs based on the dry tons per day of system capacity. The O&M costs included electricity, natural gas for startup, chemicals, operation labor, and maintenance materials. The annual O&M costs for small systems (up to 10 dtpd) are shown in Figure 7-6.

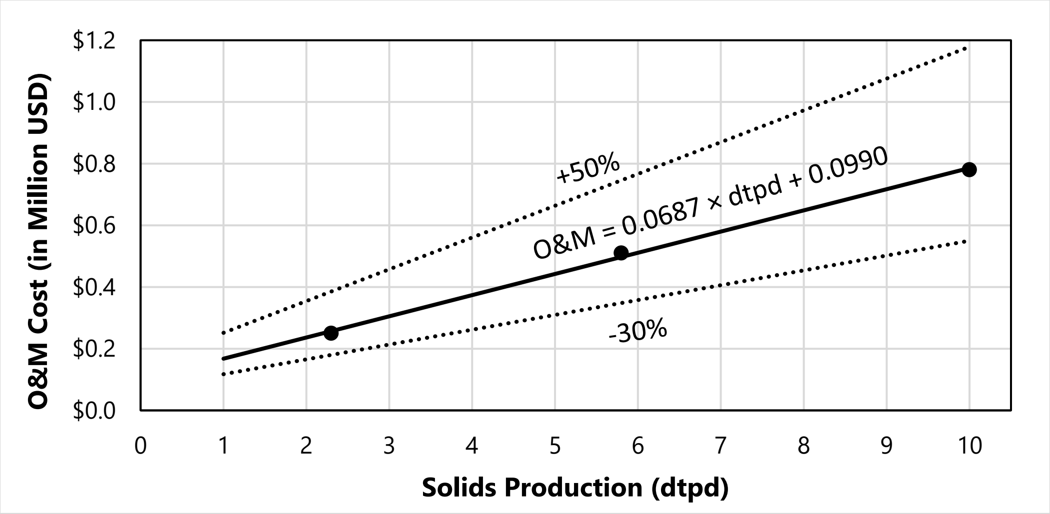


Figure 7-6 **O&M costs for biosolids pyrolysis/gasification with thermal oxidation**

Gasification and pyrolysis are well-established technologies for biowaste conversion to energy. However, using them for biosolids processing is emerging. There are several vendors commercially marketing various configurations of each process. The main obstacle these technologies need to overcome is the lack of a long-term, sustainable operation record. Other primary uncertainties for pyrolysis/gasification with thermal oxidation for biosolids include the following:

- Energy efficiency and balance, which is vendor-specific
- Documented destruction of PFAS during the process, mainly documenting PFAS destruction in the syngas at full-scale biosolids processing operations
- Required maintenance due to lack of long-term operating history of these systems
- Use and value of end products; the biochar market is not yet well developed

7.4 PFAS Removal Performance and Reliability

Both pyrolysis/gasification and SCWO are maturing processes with limited full-scale operations. However, various studies of PFAS in pyrolysis/gasification products (biochar) have shown non-detect PFAS concentrations (Davis 2020, Thoma et al. 2022). Limited data are available for the air phase from the pyrolysis/gasification; however, the process would be coupled with thermal oxidation, which has been shown to destroy PFAS at temperatures greater than 1,000°C. SCWO has been shown to destroy PFAS at a lab scale but does not have full-scale operational data (at the time this study was produced).

7.5 Example Pretreatment for PFAS Destruction Technologies Applied to Biosolids

The water content of stabilized biosolids must be reduced prior to any PFAS destruction alternatives for biosolids. The target solids percentage is dictated by the specific destruction technology. Water removed from the biosolids in dewatering and any equipment wash water would be collected and returned to the liquid wastewater treatment process.

7.5.1 Example Pretreatment for SCWO

Biosolids fed to a SCWO system must be dewatered to 12–15% total solids. Biosolids would also need to pass through a strain press upstream of dewatering to protect the SCWO unit from damage by grit, rags, or other non-sludge materials. Dewatering to this range could be achieved by any of several commercially available sludge dewatering technologies. For this study, a screw press was used to achieve a sludge concentration of 15% total solids.

The cost estimates developed in this Report for dewatering of anaerobically digested biosolids to produce 15% solids material for SCWO included the following additions:

- Biosolids storage and equalization tank
- Strain press feed pumps
- Strain press
- Screw press feed pumps
- Screw press(es)
- Dewatered biosolids pump or conveyor
- Polymer blending and feed system
- Ancillary systems, including:
    - Wash water supply
    - Ventilation for heating, cooling, and odor control
    - Supporting electrical, instrumentation, and controls
- Enclosed building
- Sitework

The number and size of major process equipment items needed will depend on the volume of biosolids produced at the WRRF and the frequency of biosolids processing. Some facilities may process biosolids on weekdays and only during a dayshift, whereas others may operate constantly. Redundancy requirements of major equipment items will depend on the facility's operating strategy. For this study, it was assumed that the pretreatment facility would constantly operate, with storage and equipment redundancy to support that operation. One spare strain press, screw press, dewatered biosolids conveyor, and polymer blending unit are included in the cost estimate. The Class 5 (AACE) cost curve (+100%/-50%) for capital costs for the dewatering system is provided in Figure 7-7.

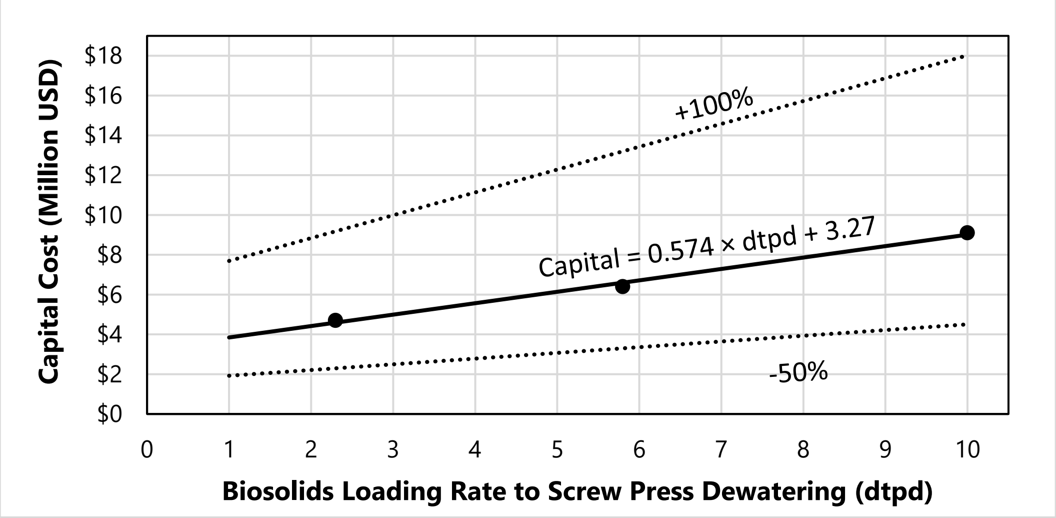


Figure 7-7 Capital cost curve for screw press dewatering to 15% solids for SCWO

7.5.2 Example Pretreatment for Pyrolysis/Gasification

Wastewater solids must be dried to reduce the water content prior to thermal drying and subsequent pyrolysis/gasification. Then the dried product is fed to the pyrolysis/gasification unit. Biosolids fed to a pyrolysis/gasification system should be dewatered as much as possible. Dewatering centrifuges are used to produce the driest dewatered biosolids compared to other biosolids dewatering technologies. Centrifuges can produce dewatered biosolids of 22%–30% total solids by mass, depending on the biosolids' characteristics and the polymer dose. For this study, dewatered biosolids were assumed to have 25% total solids.

The cost estimate of biosolids dewatering to 25% of total solids developed for this Report included the following additions:

- Dewatering feed pumps
- Dewatering centrifuge(s)
- Dewatered biosolids hopper
- Polymer blending and feed system
- Dewatered biosolids pump or conveyor
- Ancillary systems, including:
  - Wash water supply
  - Ventilation for heating, cooling, and odor control
  - Supporting electrical, instrumentation, and controls
- Enclosed building

- Sitework

The number and size of major process equipment items needed will depend on the mass and volume of biosolids produced at the WRRF and the frequency of biosolids processing. Some facilities may process biosolids on weekdays and only during a dayshift, whereas others may operate constantly. Redundancy requirements of major equipment items will depend on the facility's operating strategy. For this study, it was assumed that the pretreatment facility would constantly operate, with storage and equipment redundancy to support that operation. One spare dewatering centrifuge, dewatered biosolids conveyor, and polymer blending and feed unit are included in the cost estimate. The Class 5 (AACE) cost curve (+100%/-50%) for capital costs for the centrifuge dewatering system is provided in Figure 7-8.

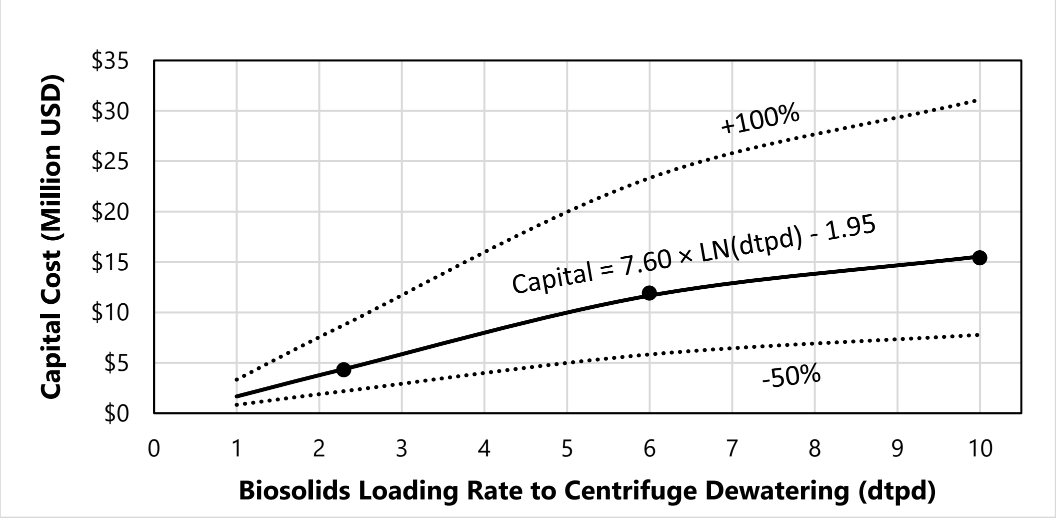


Figure 7-8 **Capital cost curve for dewatering to 25% solids for pyrolysis/gasification**

7.6 Other Considerations for Municipal Wastewater Biosolids

Treating biosolids with pyrolysis/gasification or SCWO would be a major shift away from the typical processing practices in Minnesota. Land application of Class B biosolids has been the common method of biosolids management for decades. Pyrolysis/gasification and SCWO produce a product of higher value than a Class B product. The solids drying step in the pyrolysis/gasification system produces a Class A product. Any biochar produced in the process could be land-applied as a Class A material. Operating a complex pyrolysis/gasification or SCWO system may be very challenging for some utilities. Utilities may also be concerned about the safety of these high-temperature and high-pressure systems, which may also require different operator training. Other considerations for destroying PFAS in biosolids include the following:

- The beneficial use of minerals remaining after SCWO or other management of resulting brine has not been determined. More work is needed to guide utilities to select end uses for the products of these treatment processes.

- Pyrolysis/gasification relies on intensive heat energy and does not require the presence of oxygen to "crack" or chemically decompose organic materials. Through heat application, pyrolysis/gasification disintegrates the long hydrocarbon bonds of the incoming feed materials. Pyrolysis/gasification generates not only tars, oils, and particulate matter but also reduced sulfur and nitrogen compounds and hazardous air pollutants, including polycyclic aromatic hydrocarbons. Hence, pyrolysis/gasification technologies are both subject to rules and regulations. The U.S. EPA is actively developing regulations under the Clean Air Act Section 129 pertaining to pyrolysis and gasification units used to convert solid and semi-solid feedstocks, including municipal sludge, to make useful products such as energy, fuels, and chemical commodities. The permitting of a new facility could take significant time; time frames can vary depending on the site.

8 Mixed MSW Landfill Leachate

8.1 Project Assumptions for Mixed MSW Landfill Leachate

8.1.1 Description and Scale of Required Treatment

Most MSW is managed in landfills (Kjeldsen et al. 2002). As of 2018, 63% of MSW was sent to landfills in the United States (U.S. EPA 2022b). Landfills generate leachate mainly from rainwater percolating through the waste layers in the landfill, both during operations and after closure. This leachate is generally collected for management, which can include recirculation for moisture compaction at the working face or within the waste mass, direct discharge to a WRRF, pretreatment with discharge to a WRRF, or land application. In Minnesota, mixed MSW landfills primarily discharge collected leachate to centralized municipal WRRFs for treatment or other waste management. Treatment options and cost curves for mixed MSW landfill leachate will be evaluated for facilities ranging from 1 gpm to 100 gpm, approximately 1/100th of the scale for municipal WRRF effluent considered in this study.

For this Report, the primary goal was to evaluate PFAS treatment alternatives for their ability to separate and mineralize PFAS from mixed MSW landfill leachate. The treated landfill leachate could still be sent to a WRRF for final treatment and disposal before discharging to surface water. However, in some cases, the treatment alternatives considered in this Report may sufficiently treat the landfill leachate to allow discharge to surface water instead of continuing to discharge to a WRRF. This should be evaluated on a site-specific basis.

8.1.2 PFAS Influent Concentrations and Treatment Targets

PFAS-containing products, like food packaging, clothing, and non-stick cookware, are the source of the PFAS in landfill leachate (Berg et al. 2022). Longer-chain PFAS commonly remain attached to solids within the landfill, while short-chain PFAS mobilize to the liquid leachate phase. As a result, landfill leachate is frequently enriched in short-chain PFCAs (Wei, Xu, and Zhao 2019). Other types of PFAS most relevant in mixed MSW landfill leachate are long-chain PFAAs, fluorotelomers, and precursors. Target PFAS selection was detailed in Section 2.3. Typical and high concentrations assumed as a design basis were developed from available literature and are shown in Table 8-1. These values are not intended to reflect the entire range of PFAS concentrations that may be present in all landfill leachates.

Table 8-1 Assumed influent concentrations and treatment goals for target PFAS for mixed MSW landfill leachate (all units in ng/L)

PFAS	Typical Concentrations[1]	High Concentrations[1]	Treatment Target
PFBA	950	2,600	5
PFBS	250	650	5
PFHxA	1,500	4,000	5
PFHxS	350	750	5
PFOA	900	1,900	5
PFOS	150	300	5
6:2 FTS	150	350	5
N-EtFOSAA	150	450	5

[1] Data from Lang et al.; 2017 data on 87 samples from 18 landfills across the U.S. Typical concentrations reflect the mean of 87 samples, and high concentrations reflect the 90th percentile of 87 samples. These target PFAS generally represented between 60% and 85% of the sum of PFAS analyzed. Values were rounded to the nearest 50 ng/L and two significant digits.

8.1.3 Water Quality Assumptions for Pretreated Mixed MSW Landfill Leachate Quality

Non-PFAS water quality parameters have significant implications for PFAS pretreatment and treatment of landfill leachate. The impacts of these water quality parameters are typically magnified in landfill leachate, considering its complexity and elevated concentrations, compared to WRRFs. For the pretreatment and cost analysis, it is essential to define non-PFAS parameters for landfill leachate.

Landfill leachate quality is highly variable and dependent on waste source, landfill age, and landfill type (Kjeldsen et al. 2002). In general, landfill leachate is a high-strength waste, with 20 to 57,000 mg/L of BOD_5 and 2,000 to 60,000 mg/L total solids (TS) (Kjeldsen et al. 2002). It also typically has higher concentrations of ammonia, salts, and organic carbon than municipal wastewater. Landfill leachate has a very low oxidation state, reflecting anaerobic conditions within the landfill. As a result, metals and nutrients are expected to be present in their most reduced forms (e.g., nitrogen present as ammonia and metals as dissolved metals).

Typical mixed MSW landfill leachate concentrations are listed in Table 8-2. This study chose two landfill leachate qualities for non-PFAS-related parameters meant to be representative of typical mixed MSW landfill leachate quality in Minnesota. The first landfill leachate quality is intended to represent raw mixed MSW landfill leachate for the alternatives not requiring pretreatment (Alternatives 8a and 8b). The second is for alternatives requiring pretreatment before PFAS removal (Alternatives 1a, 1b, 5a, 5b, and 7b). It reflects assumed water quality before PFAS separation technologies based on equipment fouling potential outlined previously in Section 5.2. One example of a pretreatment train and potential costs to provide this water quality at a landfill site is discussed in Section 8.5.

Table 8-2 Assumed initial and pretreated mixed MSW landfill leachate quality

Parameter	Units	Assumed Water Quality in Untreated Mixed MSW Landfill Leachate			Assumed Concentration in Pretreated Water (Influent for all Other PFAS Management Alts Evaluated)
		Typical Value (Assumed Influent Water Quality for Alt 8a and 8b)	Minimum Value[1]	Maximum Value[1]	
pH	SU	8[1]	4.5	9	8
TDS	mg/L	6,300[2]	Not reported	Not reported	6,300
TSS	mg/L	3,000[2]	2,000[3]	60,000[3]	1[4]
COD	mg/L	3,100[5]	140	152,000	90[6]
BOD$_5$	mg/L	850[5]	20	57,000	60[4]
TOC	mg/L	Not reported	30	29,000	2[4]
Alkalinity	mg/L as CaCO$_3$	6,900[2]	Not reported	Not reported	6,900
Chloride	mg/L	1,800[5]	150	4,500	1,800
Sulfate	mg/L	200[1]	Not reported	Not reported	200
Sodium	mg/L	1,600[5]	Not reported	Not reported	1,600
Potassium	mg/L	1,100[1]	Not reported	Not reported	1,100
Calcium	mg/L	110[1]	10	72,000	110
Magnesium	mg/L	150[1]	30	15,000	150
Ammonia	mg/L as N	1,200[1]	50	2,200	1,100[7]
Phosphorus	mg/L	10[1]	0.1	23	5[8]
Iron	µg/L	22,000[5]	3,000	5,500,000	0.1[4]
Manganese	µg/L	1,800[5]	30	1,400,000	0.1[4]

[1] Based on a landfill leachate review article (Kjeldsen et al. 2002).

[2] Provided by the MPCA.

[3] Kjeldsen et al. provides minimum and maximum concentrations for total solids, not TSS. The range of total solids is shown here.

[4] Based on the pretreatment discussion in Section 5.2.

[5] Based on Vermont landfill leachate study (Brown & Caldwell 2019).

[6] Based on expected BOD$_5$ reduction due to pretreatment.

[7] Ammonia is expected to decrease through biological pretreatment, however, a majority of the ammonia would remain due to the pH adjustment to prevent toxicity.

[8] Phosphorus is expected to decrease due to biological uptake.

8.2 PFAS Management Alternatives Evaluation Results

Each PFAS management alternative identified for water-phase PFAS separation and destruction in Section 4.1 was scored for each evaluation criterion described in Section 5.1 for mixed MSW landfill leachate.

8.2.1 Evaluation Scoring Results

Alternatives evaluated for mixed MSW landfill leachate, as described in Section 4.1, were:

- Alternative 1a: GAC sorption followed by off-site, high-temperature incineration of GAC

- Alternative 1b: RO membrane separation followed by GAC sorption of concentrate and off-site, high-temperature incineration of GAC
- Alternative 5a: modified clay media sorption followed by off-site, high-temperature incineration of modified clay
- Alternative 5b: RO membrane separation followed by modified clay media sorption of concentrate and off-site, high-temperature incineration of modified clay
- Alternative 7b: RO membrane separation followed by SCWO of concentrate
- Alternative 8a: foam fractionation followed by off-site, high-temperature incineration of foamate
- Alternative 8b: foam fractionation followed by SCWO of foamate

Evaluation results are summarized in Table 8-3 and described in subsequent sections. Specific criteria and subcriteria were previously defined in Section 5.1. Based on the screening, Alternatives 1a, 1b, 5a, and 8a were retained for preliminary design and cost curve development.

Table 8-3 Alternatives evaluation results for mixed MSW landfill leachate

Alternative Number		**1a**	**1b**	**5a**	**5b**	**7b**	**8a**	**8b**
Membrane separation technology		-	RO	-	RO	RO	FF	FF
PFAS sorption technology		GAC	GAC	MC	MC	-	-	-
PFAS destruction technology		**Inc**	**Inc**	**Inc**	**Inc**	**SCWO**	**Inc**	**SCWO**
Technical Feasibility	weight	*59*	*53*	*56*	*50*	*45*	*47*	*38*
PFAS separation efficiency	5	3	3	3	3	3	2	2
PFAS destruction efficiency	5	3	3	3	3	3	3	3
Degree of commercialization	3	3	3	2	2	1	2	1
Reliability of performance	2	1	1	1	1	3	2	2
Simplicity of operation/maintenance	4	3	2	3	2	1	2	1
Operator and public health	2	3	2	3	2	1	2	1
Economic Feasibility	**weight**	***46***	***41***	***41***	***36***	***26***	***40***	***34***
Relative capital costs	4	3	3	3	3	1	2	1
Relative operation and maintenance costs	3	3	2	3	2	1	2	3
Relative energy consumption	2	3	2	3	2	1	2	2
Relative complexity and cost of pretreatment	4	1	1	1	1	3	3	3
Applicability at scale	5	3	3	2	2	1	2	1
Byproducts Management	**weight**	***3***	***4***	***3***	***4***	***6***	***3***	***5***
Beneficial reuse opportunity for water or byproducts	1	1	2	1	2	2	1	1
Potential for media shifting of PFAS	2	1	1	1	1	2	1	2
Total Score		**108**	**98**	**100**	**90**	**77**	**93**	**74**
Retained?		**x**	**x**	**x**			**x**	

RO = RO membrane separation, GAC = granular activated carbon, MC = modified clay, Inc = high-temperature incineration, SCWO = supercritical water oxidation, FF = foam fractionation

8.2.2 Technical Feasibility

All sorption media alternatives evaluated received a score of 3 for **PFAS separation efficiency** because they were chosen to remove all target PFAS under selected operating conditions. While short-chain PFAS such as PFBA may break through quickly in these systems, treated water could potentially be kept below treatment targets given frequent monitoring and proactive changeout. RO/SCWO also received a score of 3 in this category because RO permeate and SCWO condensate routed to effluent are both expected to contain very low concentrations of PFAS. Alternatives with foam fractionation as the primary technology scored a 2 because foam fractionation is not currently effective at removing short-chain PFAS.

All alternatives received a score of 3 for **PFAS destruction efficiency** because both high-temperature incineration and SCWO remove 99% or more of PFAS. However, both have the potential for additional PFAS formation (Krause et al. 2022; L. Winchell et al. 2022). Additional research is needed for both these technologies to demonstrate the degree of mineralization achievable and the nature of byproducts formed.

Alternatives including combinations of RO membrane separation, GAC, and high-temperature incineration received a score of 3 for **degree of commercialization**. All of these technologies have been widely applied in multiple industries for PFAS removal. Alternatives including modified clay and foam fractionation received a score of 2 because they are commercially applied but in fewer industries. Alternatives including SCWO received a score of 1 because the technology is still developing for PFAS removal, especially in the water phase. It is currently limited to pilot-scale installations.

Sorption media received a score of 1 for **reliability of performance** because rapid breakthrough of short-chain PFAS could result in detectable concentrations in treated effluent, depending on monitoring frequency and mobilization time for changeout. Flow-rate variation in media vessel influent can also adversely affect the EBCT and associated mass transfer zone in sorption reactors, which can cause premature breakthrough. Alternatives with foam fractionation received a 2, with higher reliability for long-chain PFAS but low reliability for short-chain PFAS. RO/SCWO received a score of 3 because treatment efficiency is not impacted by changing water quality, flows, or environmental conditions.

Alternatives with sorption media combined with high-temperature incineration received the highest score of 3 for **simplicity of operation**. Alternatives with RO and media received a score of 2 due to the operational complexity of running an RO system, including system monitoring and membrane cleaning and replacement. Foam fractionation with high-temperature incineration received a score of 2 because it is expected that foam fractionation systems are more complex to operate than sorption media systems. Foam fractionation combined with SCWO received a score of 1 due to the complexity of operating both technologies.

Sorption media alternatives with high-temperature incineration received a score of 3 for **operator and public health** because there are minimal risks associated with these types of treatment and destruction technologies. Alternatives with RO/SCWO and foam fractionation with SCWO were scored a 1 due to the combined risks of high-temperature SCWO, high-pressure RO, and the PFAS volatilization potential associated with foam fractionation. All other alternatives received a score of 2 because although they are used in the industry, there is an increased risk to operators and public health due to one or more of the following: addition of chemicals, high operating temperatures, or high operating pressures.

8.2.3 Economic Feasibility

Alternatives that combine two high-energy technologies, like RO/SCWO and foam fractionation with SCWO have the highest **relative capital cost** and therefore received a score of 1. Foam fractionation with off-site, high-temperature incineration received a score of 2 due to lower on-site capital expenses. Alternatives with sorption media received a score of 3, reflecting the lowest relative capital cost. While using RO as a pre-concentration step adds capital costs, these are expected to be offset by the lower capital cost of media vessels treating only concentrate relative to alternatives with media vessels treating the full leachate flow.

When considering **relative O&M costs (without energy)**, alternatives with single-use media disposed of via high-temperature incineration received a score of 3, with primary O&M costs associated with transport and off-site incineration of sorption media. Sorption media alternatives with RO upstream received a score of 2 due to increased operational and maintenance costs of the added RO process. Foam fractionation is less expensive than RO and produces a lower volume of foamate than RO concentrate (higher recovery), resulting in lowered O&M costs and a score of 3 when paired with high-temperature incineration destruction and 2 with SCWO destruction. RO/SCWO is expected to have the highest operational costs and received a score of 1.

Sorption media without RO pre-concentration is expected to have the lowest **relative energy consumption**, with a score of 3. Each alternative with RO and sorption media for concentrate treatment and high-temperature incineration of the media scored a 2. RO/SCWO received a score of 1 due to the high energy required for the high temperatures and pressure needed to treat the full RO concentrate stream. Foam fractionation with SCWO scored a 2 due to the smaller waste stream routed to the SCWO feed. Foam fractionation with off-site high-temperature incineration also scored a 2, assuming similar energy use for SCWO and incineration of the same volume of liquid.

Alternatives using GAC, modified clay, or RO scored a 1 for **relative complexity and cost of pretreatment** because all would require pretreatment, as described in Section 8.1.38.5. Foam fractionation alternatives scored a 3 because they do not require complex pretreatment. RO/SCWO also scored a 3, assuming that specialty wide-spaced RO could be applied without pretreatment.

Applicability at scale was somewhat pre-screened as part of technology screening. GAC and RO membrane separation have been widely applied at the scale for landfill leachate treatment (1–100 gpm), so alternatives including only these separation technologies received a score of 3. Modified clay media sorption and foam fractionation have limited applications at this flow rate, and alternatives including these technologies received a score of 2. Alternatives with SCWO scored a 1 due to limited commercial application for PFAS management.

8.2.4 Byproducts Management

Alternatives with two opportunities for **beneficial reuse** of the treated waste stream or byproducts received a score of 3. Alternatives with one form of beneficial reuse received a score of 2. Alternatives with no potential beneficial reuse received a score of 1. For mixed MSW landfill leachate, no alternatives included two opportunities for reuse, and the highest score was a 2. The one example of reuse was for alternatives with RO, which produces RO permeate that could be reused on-site at the landfill (i.e., for dust control or compaction).

All alternatives with high-temperature incineration received a score of 1 for **potential media shifting of PFAS.** This is due to the current uncertainty and ongoing industry research investigating PFAS transformations and emissions associated with high-temperature incineration. Alternatives with SCWO destruction received a score of 2 because the possibility of PFAS media shifting is unknown.

8.2.5 PFAS Management Alternatives for Preliminary Design

Based on the screening described in Section 8.2, four PFAS management alternatives were carried forward to preliminary design and cost analysis for mixed MSW landfill leachate:

- Alternative 1a: GAC sorption followed by off-site, high-temperature incineration of GAC
- Alternative 1b: RO membrane separation followed by GAC sorption of concentrate and off-site, high-temperature incineration of GAC
- Alternative 5a: modified clay media sorption followed by off-site, high-temperature incineration of modified clay
- Alternative 8a: foam fractionation followed by off-site, high-temperature incineration of foamate

8.3 Alternative Preliminary Design and Cost Curve Development

8.3.1 Alternative 1a (Landfill Leachate)

For Alternative 1a, pretreated mixed MSW landfill leachate meeting the water quality listed in Table 8-2 flows directly to lead-lag GAC adsorption pressure vessels. Once GAC media is exhausted with respect to target PFAS (limited by PFBA and PFBS in this case), it would be sent to high-temperature incineration because the flows of landfill leachate considered in this Report would not meet the GAC volume required for site-specific reactivation. This process flow is summarized in Figure 8-1.

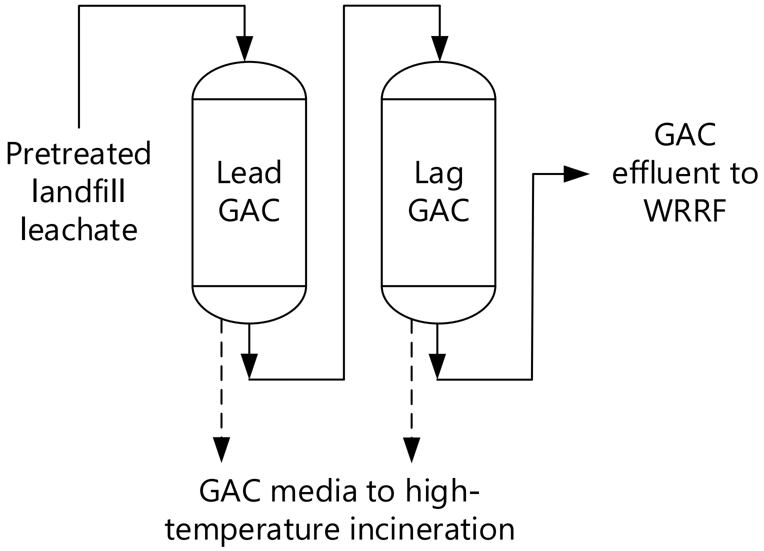


Figure 8-1 Conceptual process flow diagram for Alternative 1a for landfill leachate

Key design basis assumptions for Alternative 1a are listed in Table 8-4, with a detailed design basis in Appendix E.

Table 8-4 Summary of design basis assumptions for Alternative 1a for landfill leachate

Design Parameter	Basis		
	0.0014 MGD/1 gpm	0.014 MGD/10 gpm	0.14 MGD/100 gpm
Vessel capacity (lb GAC)	90	750	10,000
Number of trains	2	1	1
Number of vessels	4	2	2
EBCT per vessel (min)	15	15	15
HLR (gpm/sq. ft)	0.5	3.2	2.0
Estimated bed volumes to breakthrough[1]	5,100	5,100	5,100
GAC disposal route	High-temperature incineration	High-temperature incineration	High-temperature incineration

[1] Defined as the estimated volume of water treated through the lead media vessel until the first detection of PFAS at 5 ng/L. Described further in Section 5.3.4 and Appendix D.

Figure 8-2 and Figure 8-3 illustrate estimated capital and annual O&M costs for a range of influent flow rates. These costs do not include pretreatment costs described in Section 8.5. Detailed capital and O&M cost estimates are in Appendix E.

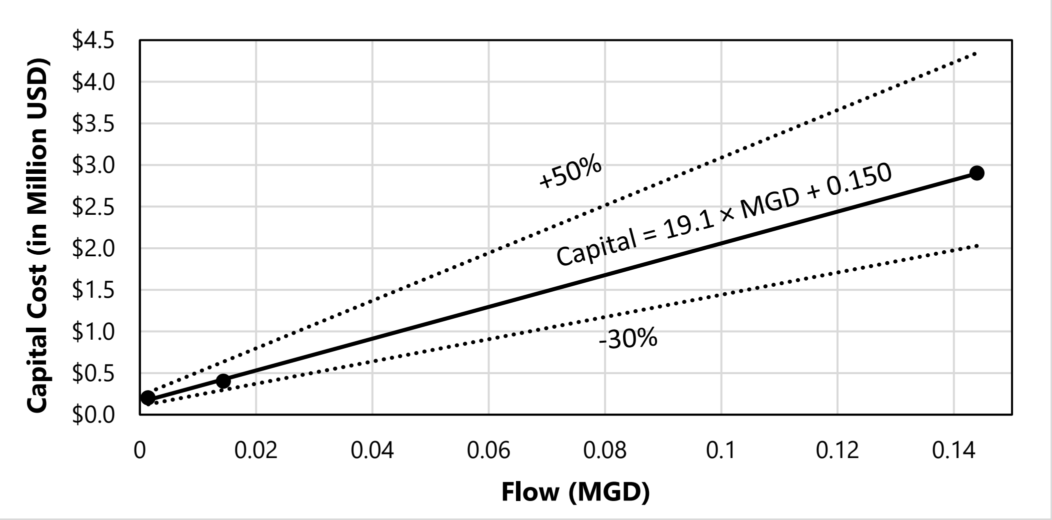


Figure 8-2 Capital cost curve for Alternative 1a for landfill leachate

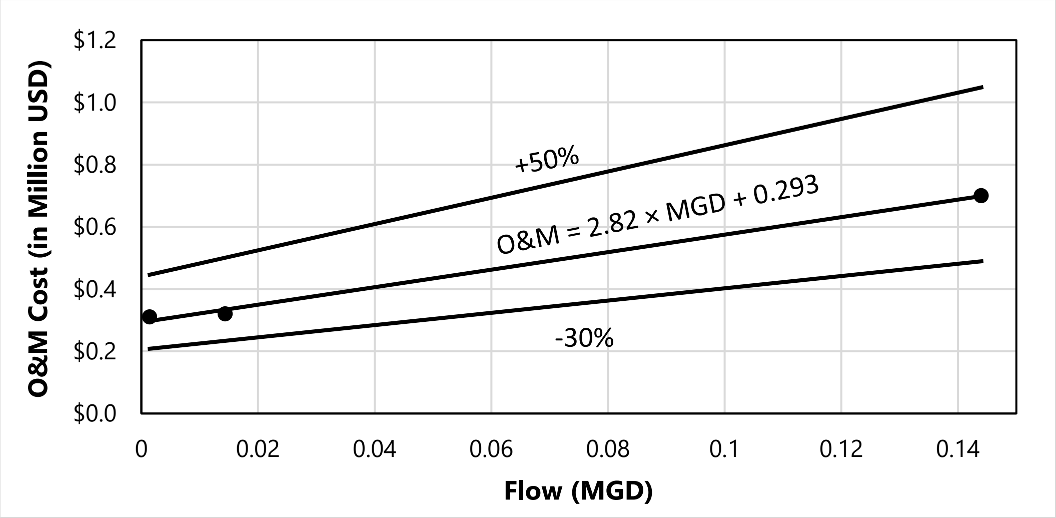


Figure 8-3 **O&M cost curve for Alternative 1a for landfill leachate**

Estimated annual O&M costs are similar to estimated capital costs at low flows but approximately one-quarter of estimated capital costs at higher flows. The highest operating costs are associated with labor and sorption media replacement and incineration.

Primary uncertainties for Alternative 1a for landfill leachate treatment include the following:

- Actual GAC breakthrough timing and requirements, depending on actual pretreated water quality
- Potential for GAC fouling limiting bed life over PFAS breakthrough
- Pretreatment processes needed and ability to meet pretreatment targets
- Location and incineration fees of a selected high-temperature incineration facility
- The capital costs for the lowest flow rates may change depending on facility preference for operational flexibility. The vessels included in this analysis are relatively small such that a typical valve tree included with larger vessels is not standard and was not included.

8.3.2 Alternative 1b (Landfill Leachate)

Alternative 1b includes GAC sorption, as previously described for Alternative 1a, with the addition of an RO membrane pre-concentration step. Pretreated mixed MSW landfill leachate meeting the water quality listed in Table 8-2 is routed directly to RO membranes. The RO concentrate flows to lead-lag GAC adsorption pressure vessels. RO permeate and GAC effluent would be combined and sent to a WRRF. Once GAC media is exhausted with respect to target PFAS (limited by PFBA and PFBS in this case), it would be sent to high-temperature incineration. This process flow is summarized in Figure 8-4.

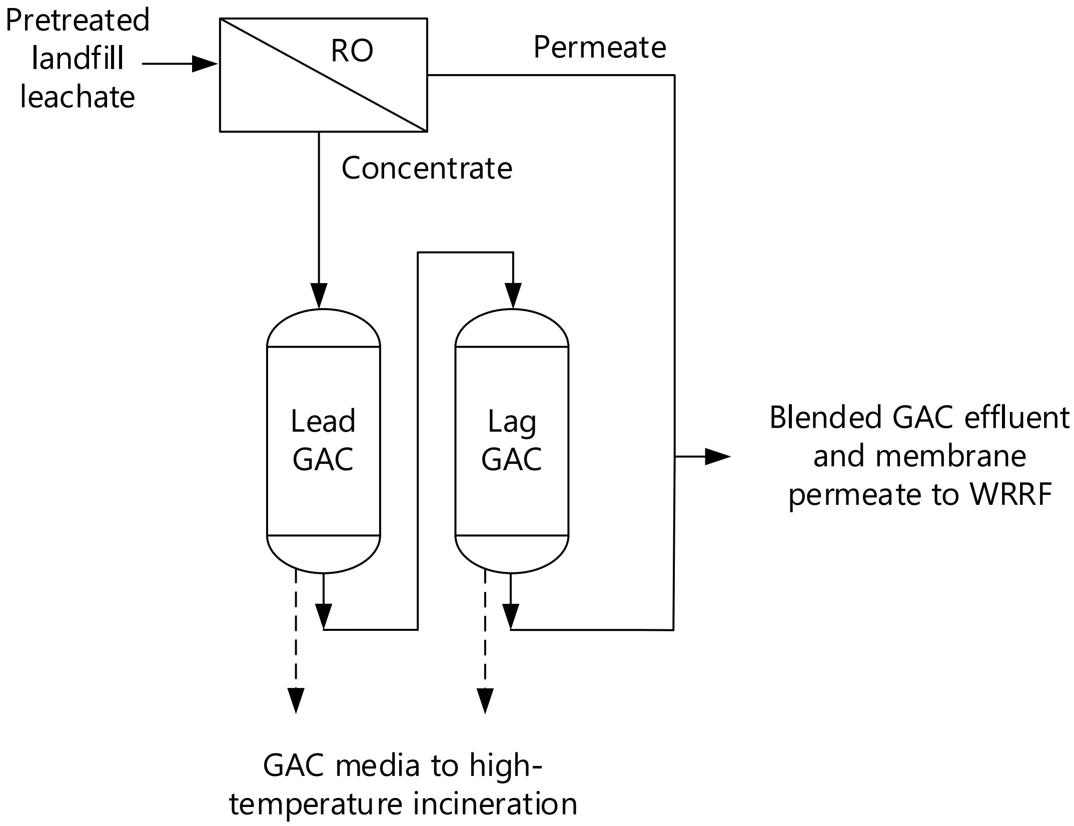


Figure 8-4 Conceptual process flow diagram for Alternative 1b for landfill leachate

Key design basis assumptions for Alternative 1b are listed in Table 8-5, with a detailed design basis in Appendix E.

Table 8-5 Summary of design basis assumptions for Alternative 1b for landfill leachate

Design Parameter	Basis		
	0.0014 MGD/1 gpm	0.014 MGD/10 gpm	0.14 MGD/100 gpm
RO			
Recovery (%)	65	65	65
Flux (gal./sq. ft/day)	16	16	16
GAC			
Vessel capacity (lb GAC)	90	250	2,000
Number of trains	2	2	2
Number of vessels	4	4	4
EBCT per vessel (min)	15	15	15
HLR (gpm/sq. ft)	0.5	0.6	1.0
Estimated bed volumes to breakthrough[1]	3,600	3,600	3,600
GAC disposal route	High-temperature incineration	High-temperature incineration	High-temperature incineration

[1] Defined as the estimated volume of water treated through the lead media vessel until the first detection of PFAS at 5 ng/L. Described further in Section 5.3.4 and Appendix D.

Figure 8-5 and Figure 8-6 illustrate estimated capital and annual O&M costs for a range of influent flow rates. These costs do not include pretreatment costs described in Section 8.5. Detailed capital and O&M cost estimates are in Appendix E.

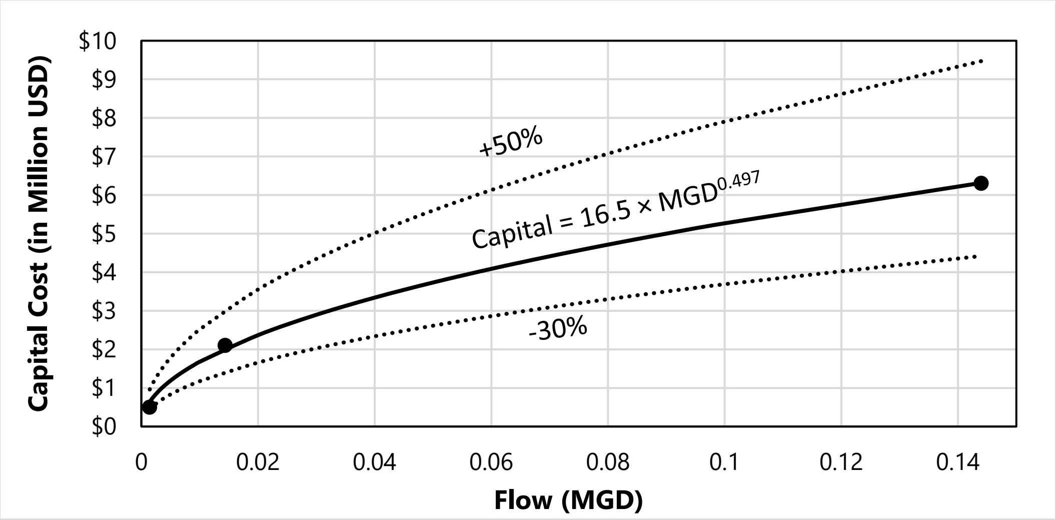


Figure 8-5 **Capital cost curve for Alternative 1b for landfill leachate**

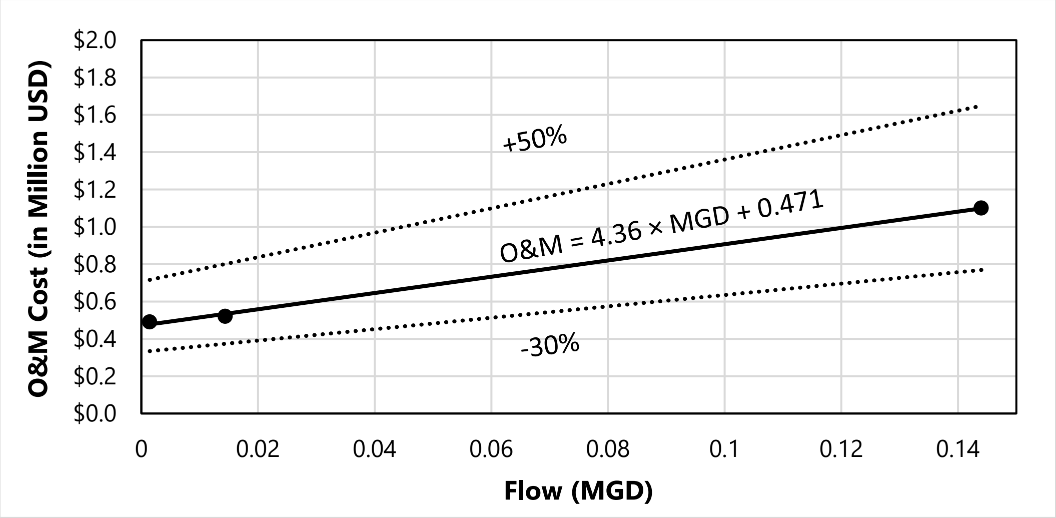


Figure 8-6 O&M cost curve for Alternative 1b for landfill leachate

Estimated capital costs are approximately twice those for Alternative 1a, and O&M costs are approximately one-and-a-half times those for Alternative 1a. The highest operating costs are associated with utilities, labor, and sorption media replacement and incineration.

Primary uncertainties for Alternative 1b for landfill leachate treatment include the following:

- Achievable recovery of RO membranes and associated energy usage
- Fouling potential of RO membranes, depending on actual pretreated water quality
- Actual GAC breakthrough timing and requirements, depending on actual pretreated water quality
- Potential for GAC fouling limiting bed life over PFAS breakthrough
- Pretreatment processes needed and ability to meet pretreatment targets
- Location and incineration fees of selected high-temperature incineration facility
- The capital costs for the lowest flow rates may change depending on facility preference for operational flexibility. The vessels included in this analysis are relatively small such that a typical valve tree included with larger vessels is not standard and was not included.

8.3.3 Alternative 5a (Landfill Leachate)

For Alternative 5a, pretreated mixed MSW landfill leachate meeting the water quality listed in Table 8-2 flows directly to lead-lag modified clay adsorption pressure vessels. Once modified clay media is exhausted with respect to the target PFAS (likely limited by breakthrough of short-chain PFAS), the spent media is sent to high-temperature incineration. This process flow is summarized in Figure 8-7.

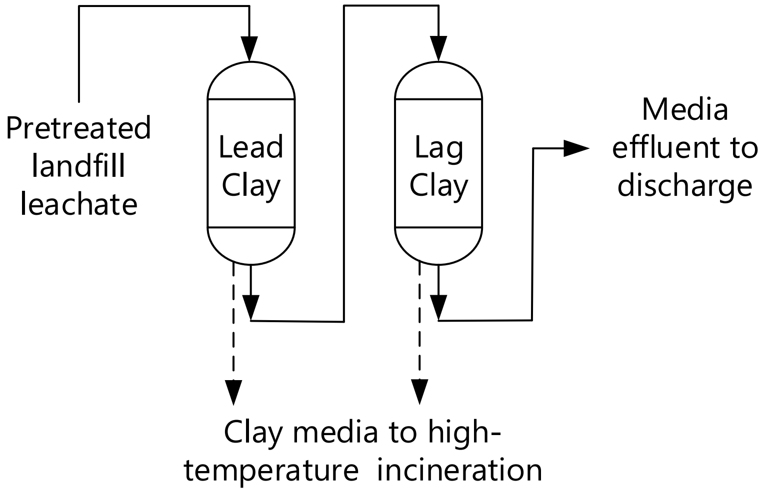


Figure 8-7 **Conceptual process flow diagram for Alternative 5a for landfill leachate**

Key design basis assumptions for Alternative 5a are listed in Table 8-6, with a detailed design basis in Appendix E.

Table 8-6 **Summary of design basis assumptions for Alternative 5a for landfill leachate**

Design Parameter	Basis		
	0.0014 MGD/1 gpm	0.014 MGD/10 gpm	0.14 MGD/100 gpm
Vessel capacity (cu. ft media)	3	5	40
Number of trains	1	3	4
Number of vessels	2	6	8
EBCT per vessel (min)	10	10	10
HLR (gpm/sq. ft)	0.9	1.9	3.5
Estimated bed volumes to breakthrough[1]	5,000	5,000	5,000
Media disposal route	High-temperature incineration	High-temperature incineration	High-temperature incineration

[1] Defined as the estimated volume of water treated through the lead media vessel until the first detection of PFAS at 5 ng/L. Described further in Section 5.3.4 and Appendix D.

Figure 8-8 and Figure 8-9 illustrate estimated capital and annual O&M costs for a range of influent flow rates. These costs do not include pretreatment costs described in Section 8.5. Detailed capital and O&M cost estimates are in Appendix E.

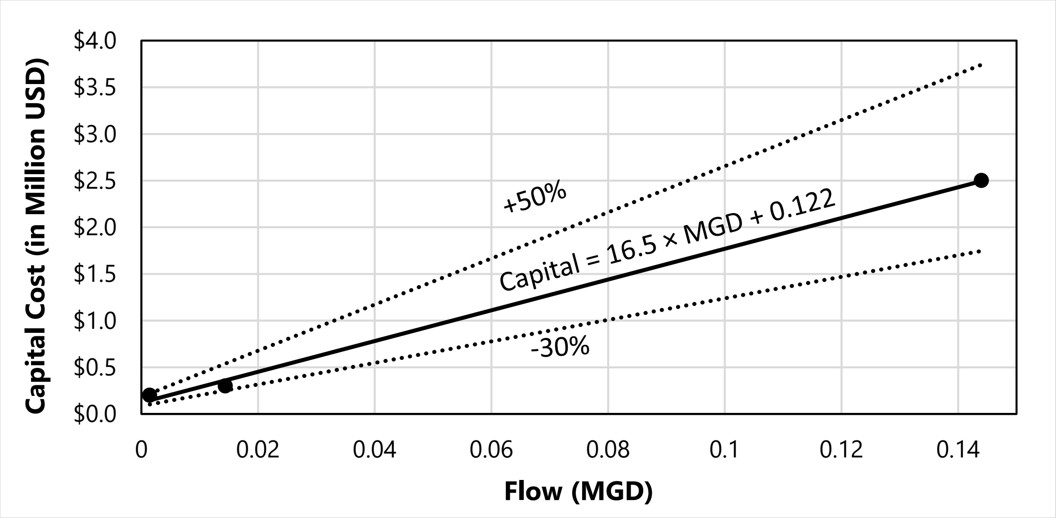


Figure 8-8 **Capital cost curve for Alternative 5a for landfill leachate**

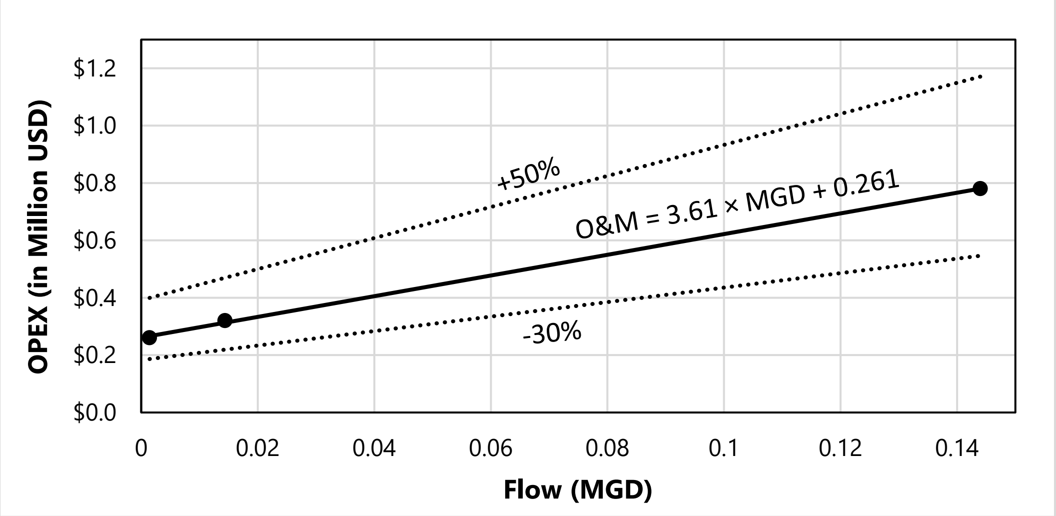


Figure 8-9 **O&M cost curve for Alternative 5a for landfill leachate**

Estimated annual O&M costs and estimated capital costs for Alternative 5a are similar to estimated costs for Alternative 1a. The highest operating costs for 5a are associated with labor and sorption media replacement and incineration.

Primary uncertainties for Alternative 5a for landfill leachate treatment include the following:

- Actual PFAS breakthrough timing through modified clay media, depending on media performance and actual pretreated water quality
- Potential for modified clay fouling to limit bed life over PFAS breakthrough
- Pretreatment processes needed and ability to meet pretreatment targets
- Location and incineration fees of a selected high-temperature incineration facility

- The capital costs for the lowest flow rates may change depending on facility preference for operational flexibility. The vessels in this analysis are relatively small such that a typical valve tree included with larger vessels is not standard and was not included.

8.3.4 Alternative 8a (Landfill Leachate)

For Alternative 8a, mixed MSW landfill leachate would be pumped to a series of foam fractionation vessels. The contact water initially goes through a bag filter to remove suspended solids, avoiding solids buildup in the foam fractionation vessels. The foam generated from the primary foam fractionation vessel is further enriched in a secondary foam fractionation vessel. The air that generates the foam in the closed foam fractionation vessel is discharged to carbon canisters to capture volatile organics. The resulting foam is collected and transported to a high-temperature incineration facility for destruction. This process flow is summarized in Figure 8-10.

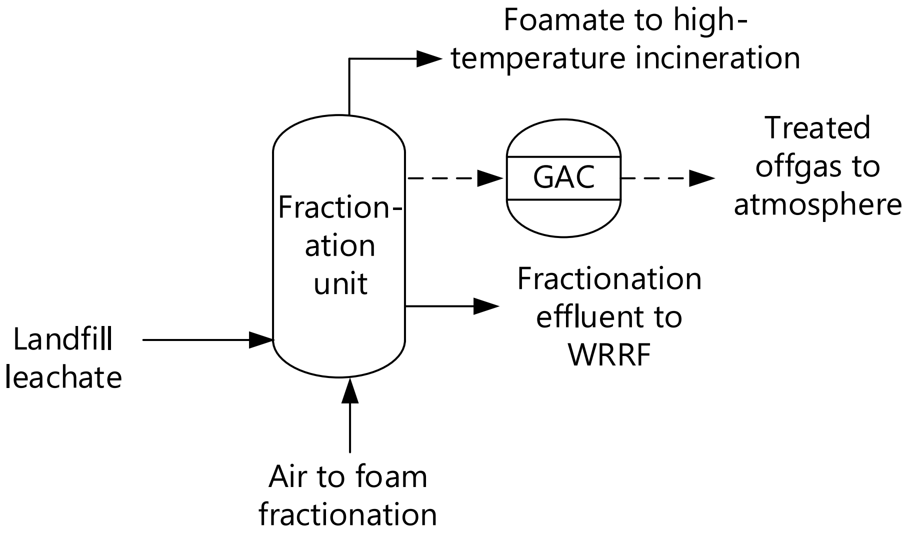


Figure 8-10 Conceptual process flow diagram for Alternative 8a for landfill leachate

Key design basis assumptions for Alternative 8a are listed in Table 8-7, with a detailed design basis in Appendix E.

Table 8-7 Summary of design basis assumptions for Alternative 8a for landfill leachate

Design Parameter	Basis		
	0.0014 MGD/1 gpm	0.014 MGD/10 gpm	0.14 MGD/100 gpm
Number of primary vessels	2	2	4
Number of secondary vessels	1	1	2
Assumed concentration factor[1]	1,000,000	1,000,000	1,000,000
Expected foamate flow rate (gpd)	0.0014	0.014	0.14

[1] Based on vendor input.

Figure 8-11 and Figure 8-12 illustrate estimated capital and annual O&M costs for a range of influent flow rates. Detailed capital and O&M cost estimates are in Appendix E.

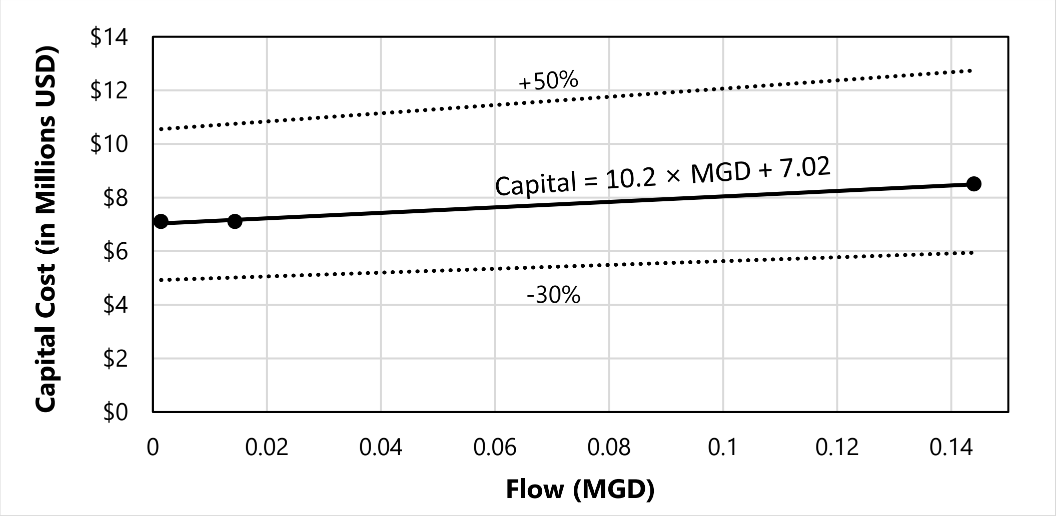


Figure 8-11 Capital cost curve for Alternative 8a for landfill leachate

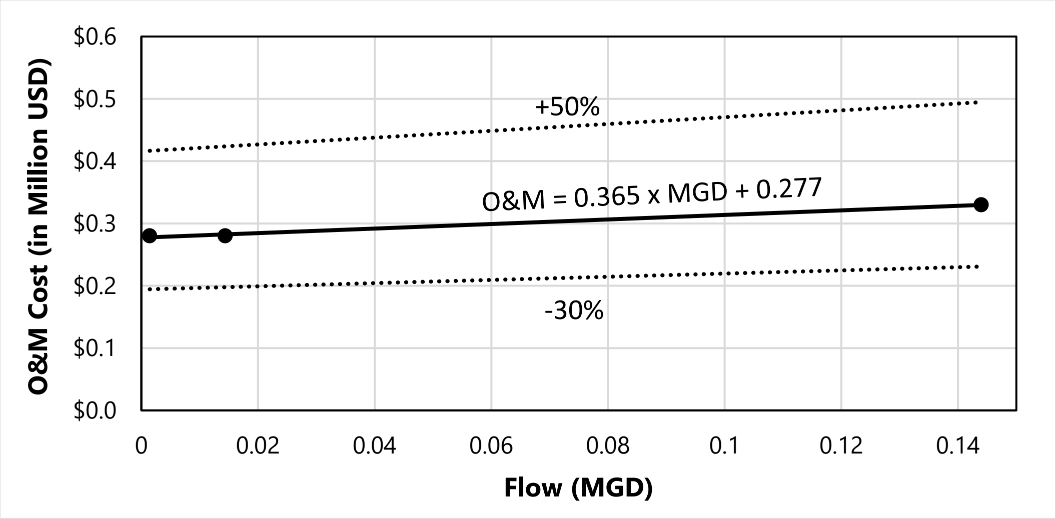


Figure 8-12 O&M cost curve for Alternative 8a for landfill leachate

Estimated annual O&M costs are approximately one order of magnitude lower than the estimated capital costs. The highest operating costs are associated with utilities, maintenance, and labor. The capital costs are an order of magnitude higher than Alternative 1a for the low and middle flow rates but the same order of magnitude for the high flow rate. The O&M costs are similar to those for Alternative 1a. However, unlike Alternative 1a, there are no O&M costs associated with pretreatment for Alternative 8a. Although pretreatment O&M costs were not considered in this report, long-term O&M may cost less for Alternative 8a and should be evaluated on a site-specific basis.

Primary uncertainties for Alternative 8a for landfill leachate treatment include the following:

- Actual concentrating factor of the foam (assumed to be 1,000,000)
- The removal efficiency of short-chain PFAS (such as PFBA) and the potential usage rate of cationic surfactants or enhancements to improve the removal efficiency. Polishing treatment (such as media sorption) on the treated water stream from foam fractionation may be needed to achieve the PFAS performance targets of this study. This would increase the capital and O&M cost estimates provided above.
- Location and incineration fees of a selected high-temperature incineration facility

8.4 PFAS Removal Performance and Reliability

The relative ability of each alternative to meet treatment goals of <5 ng/L for target PFAS is summarized in Table 8-8. Including RO with GAC (Alternative 1a versus 1b) has limited effect on PFAS removal because the PFAS removal mechanism is the same GAC adsorption process. Currently, it is uncertain if foam fractionation (Alternative 8a) can remove 90% of short-chain PFAS. If not, this would result in pass-through of PFAS such as PFBA, PFBS, PFHxA, and possibly others into the treated water phase at concentrations higher than the 5 ng/L target. Specific references for PFAS removal efficiency are detailed further in Appendix C.

Table 8-8 PFAS removal performance and reliability for mixed MSW landfill leachate alternatives

Target PFAS	Assumed Influent Concentration (ng/L)	Percent Removal to Achieve <5 ng/L	Relative Ability to Consistently Meet Targets			
			1a: GAC	1b: RO/GAC	5a: Modified Clay	8a: Foam Fractionation with Incineration
PFBA	950	>99%	Moderate	Moderate	Moderate	Low
PFBS	250	98%	Moderate	Moderate	Moderate	Low
PFHxA	1,500	>99%	Moderate	High	Moderate	Moderate
PFHxS	350	99%	High	High	High	High
PFOA	900	>99%	High	High	High	High
PFOS	150	97%	High	High	High	High
6:2 FTS	150	97%	High	High	High	High
N-EtFOSAA	150	97%	High	High	High	High

8.5 Example Pretreatment for PFAS Separation Technologies Applied to Mixed MSW Landfill Leachate

Minnesota currently has no landfills with permanent, full-scale leachate treatment beyond settling and equalization. Therefore, we expect most landfills implementing PFAS management alternatives will require significant pretreatment prior to PFAS separation and destruction.

Biological and filtration processes are required to achieve the TOC and TSS pretreatment goals required for Alternatives 1a and 1b to limit the fouling of downstream processes, as described in Section 5.2. One challenge with biological treatment in the case of landfill leachate is the potential for ammonia toxicity. For a biological process to effectively remove the BOD_5 in landfill leachate, the free ammonia concentration in the leachate needs to be lower than 30–50 mg NH_3-N/L through ammonia removal or acidification to convert all ammonia to the less toxic ionized form (Kim et al. 2008). At raw landfill leachate concentrations and pH (8 STU), free ammonia would be toxic to the aerobic treatment microbes. As a result, acid can be added to reduce pH and associated toxicity or raise the pH and strip out the ammonia ahead of biological treatment.

One potential pretreatment train for landfill leachate would include aeration for iron oxidation to the particulate form followed by a clarifier to remove oxidized iron and TSS. After settling, the pH would be adjusted down to drive the ammonia speciation to ammonium and prevent toxicity but not low enough to prevent pH inhibition. After pH adjustment, the water would flow through an MBR for TOC, BOD_5, and TSS removal.

MBR systems require consistent flows to maintain healthy biomass. Because landfill leachate flows are variable and dependent on precipitation, the treatment system will need sufficient flow equalization to maintain continuous flow during dry periods (e.g., summer and winter). It is assumed that existing landfill infrastructure could be used for flow equalization, with minor modifications as necessary. Thus, construction costs for equalization basins are not included in the capital costs shown below. If sufficient flow equalization is not practical for a specific facility, alternative treatment could be evaluated instead of MBR treatment that could be operated as needed (i.e., batch-wise).

The example conceptual process flow and capital cost estimates for mixed MSW landfill leachate pretreatment for PFAS removal are assumed to include the following:

- Aeration—to oxidize iron
- Chemical precipitation and sedimentation—to remove oxidized iron, a fraction of the TOC, and fine suspended solids
- Acid addition—to lower pH and prevent unionized ammonia toxicity during biological treatment
- MBR—to remove BOD_5, TOC, and residual suspended solids

The conceptual pretreatment process flow is shown in Figure 8-13.

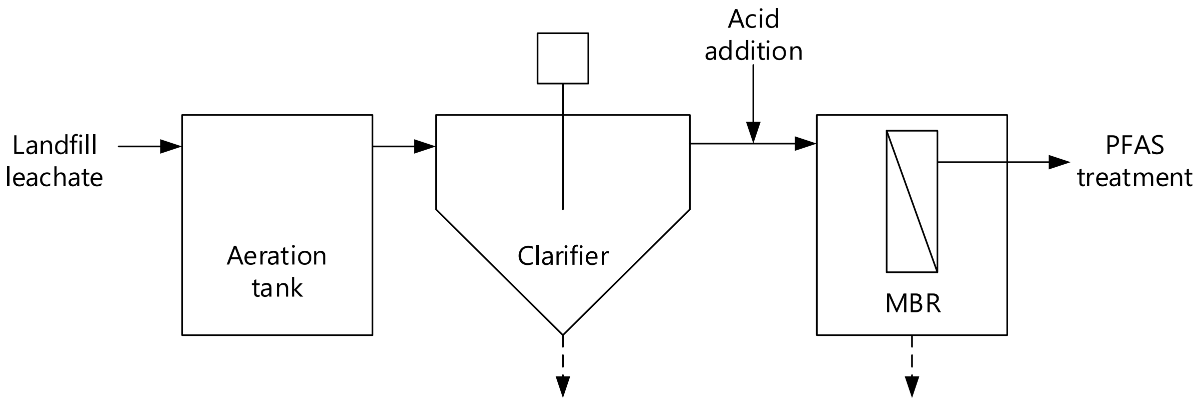
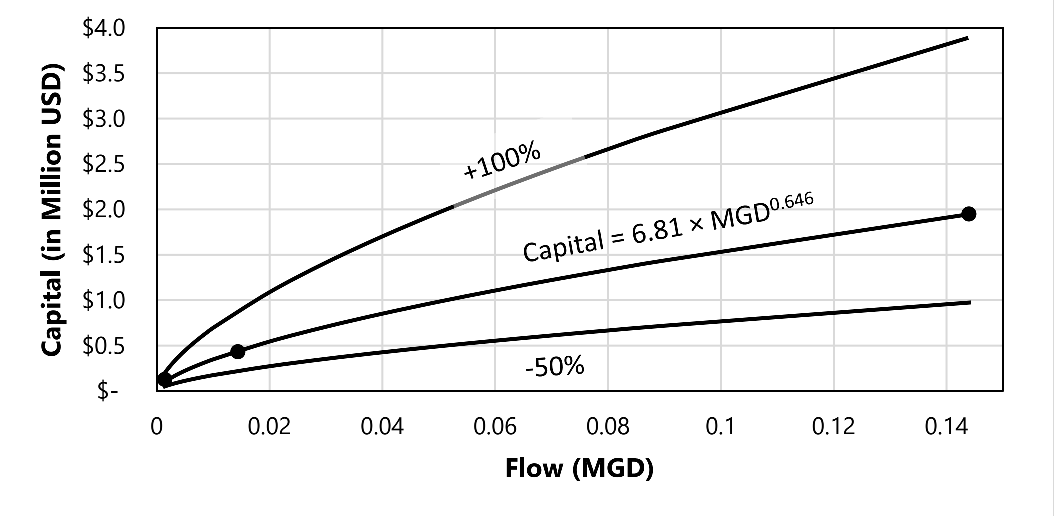


Figure 8-13 Conceptual process flow diagram for landfill leachate pretreatment

The pretreatment approach and estimated costs are meant to reflect one example but will not be appropriate for all systems. The evaluation and selection of a pretreatment system should be completed specifically for each individual site looking to implement a PFAS management approach. For example, this pretreatment example assumes relatively constant leachate flow, which might not be the case for each site. This Report evaluates the use of a clarifier for particulate iron and solids removal instead of a pond. This enables estimation of a conservative pretreatment cost due to the clarifier and associated equipment costs and the increased building footprint. If a pond exists on site that could be repurposed for solids removal, the pretreatment cost could be lower than estimated here. An additional consideration not included in this evaluation is air controls for potential volatilization of PFAS in the pretreatment units.

Capital cost estimates for this pretreatment train are shown in MBR costs from Lo, McAdam, and Judd (2015). Other equipment costs estimated from CapdetWorks (Hydromantis, v4.0). The solid line reflects the estimated cost curve. The dashed lines reflect the
+100%/-50% accuracy range. All costs scaled to November 2022 using ENR CCI (13175).

Figure 8-14. Costs are Class 5 (AACE) cost estimates with an accuracy range of +100%/-50%. This cost curve should only be used for the flow range shown (1–100 gpm; 0.0014–0.14 MGD).



MBR costs from Lo, McAdam, and Judd (2015). Other equipment costs estimated from CapdetWorks (Hydromantis, v4.0). The solid line reflects the estimated cost curve. The dashed lines reflect the +100%/-50% accuracy range. All costs scaled to November 2022 using ENR CCI (13175).

Figure 8-14 Capital cost curve for new MBR installations with pre-aeration, sedimentation, and chemical addition for landfill leachate pretreatment

The conceptual pretreatment cost estimates did not include a detailed feasibility study or any design for site-specific requirements. Pretreatment needs are expected to vary by facility due to differences in existing infrastructure, water quality, space availability, and other site-specific constraints. Thus, the degree of detail included in this Report for pretreatment is less than that provided for PFAS treatment alternatives in subsequent sections. The conceptual pretreatment costs shown are intended to provide an order-of-magnitude estimate for potential pretreatment costs. **Facilities looking to implement PFAS treatment should conduct site-specific treatment evaluations prior to capital planning or implementation.**

8.6 Other Considerations for Mixed MSW Landfill Leachate

The PFAS management alternative most appropriate for a given site will depend on space constraints, initial water quality, operational and treatment targets, and permit conditions. Sites targeting only long-chain PFAS will likely find foam fractionation (as included in Alternative 8a) the most cost-effective separation technology because it does not require pretreatment. However, foam fractionation's efficacy for separating short-chain PFAS remains limited at this time. Large landfills with short-chain removal requirements and space constraints may benefit from using conventional RO (with appropriate pretreatment) or specialty wide-spaced RO (as included in Alternative 1b) as a pre-concentration step before media sorption vessels for concentrate management. Modified clay media (Alternative 5a) are still being vetted as a sorption media for PFAS separation and may have specific advantages over GAC (Alternative 1a), such as lower capital cost and footprint.

The decision to select one of these alternatives and implement them on-site will vary site by site and based on the PFAS management strategy of the WRRF receiving the mixed MSW landfill leachate discharge. PFAS management strategies for landfill leachate should consider if on-site treatment is necessary. For example, if a downstream WRRF is implementing PFAS treatment, it may not make sense to also implement PFAS treatment at a landfill. Other considerations for removing and destroying PFAS from landfill leachate include the following:

- Off-site treatment at a regional facility may be the preferred alternative, depending on the flow and the proximity to a regional facility. If on-site treatment is selected, the feasibility of treatment to surface water standards with direct discharge should also be considered.
- Depending on the landfill leachate strength, leachate treatment with SCWO could be more favorable than for the design landfill leachate considered in this Report. Landfill leachate containing higher-strength organics may require less co-fuel when using SCWO, which could lower the operating costs for this technology. Additionally, landfill gas could be captured and cleaned to use as a co-fuel source if compressed to a liquid to potentially reduce operating costs.
- Media shifting should be evaluated, particularly when using pretreatment with aeration. Aeration may cause PFAS volatilization and result in surface deposition of PFAS. Any treatment, including aeration, should consider air pollution control strategies to prevent media shifting and contamination due to airborne PFAS.
- In addition to the technologies included in this alternatives analysis, there are a few technologies currently available at the pilot scale that might be amenable to PFAS treatment without pretreatment. Advanced electrochemical oxidation and HALT treatment, depending on performance in pilot-scale tests and ultimate costs, may be other beneficial technologies to consider for treatment of landfill leachate.

9 Compost Contact Water

9.1 Project Assumptions for Compost Contact Water

9.1.1 Description and Scale of Required Treatment

Compost contact water (also known as compost leachate) originates from precipitation runoff from active compost areas, moisture associated with source compost materials, and water added to maintain specific moisture contents during the composting process (Krogmann and Woyczechowski 2000). Minnesota administrative rules part 7035.2836 defines contact water from commercial composting facilities as:

> *[W]ater that has come into contact with source-separated organic material in the tipping area, source-separated organic material in the mixing area, rejects, residuals, or active compost.*

Water originating from curing or finished storage areas at composting facilities is considered stormwater (different from the definition of contact water) under Minnesota rules. It is managed with other site stormwater under industrial stormwater permits.

Compost contact water must be collected in lined ponds and managed according to Minnesota administrative rules part 7035.2815, subpart 9, which defines design and construction criteria for solid waste leachate collection systems.

This study addresses compost contact water from source-separated organic material (SSOM) facilities that accept both food and yard waste, including compostable packaging. Nine commercial SSOM composting facilities are permitted to accept food waste in Minnesota (Minnesota Pollution Control Agency 2022). Currently, no commercial composting facilities in Minnesota have active treatment systems for contact water. Instead, contact water is transported to local WRRFs for treatment on an as-needed basis, depending on the capacity of the on-site collection ponds.

Because compost contact water is partly dependent on precipitation, water generation is seasonal and depends on the size of a facility and the volume of precipitation received. Alternatives developed in this study for compost contact water treatment assume that generation for a given facility varies between 1–100 gpm (0.0014–0.14 MGD) based on typical compost contact water generation rates reported in Roy et al. (2018). This flow range estimate is not specific to Minnesota commercial composting facilities due to limited available data.

For this Report, the primary goal was to evaluate PFAS treatment alternatives for their ability to separate and mineralize PFAS from compost contact water. The treatment evaluated is based on water quality criteria required for effective PFAS treatment, not necessarily surface water discharge criteria. The recommended level of treatment for PFAS may provide sufficient water quality for surface water discharge in specific instances but should be evaluated in individual, site-specific studies.

9.1.2 PFAS Influent Concentrations and Treatment Targets

The MPCA and Wood Environmental & Infrastructure Solutions completed a study of PFAS concentrations in compost contact water in 2018 and 2019 (Wood Environment & Infrastructure Solutions Inc. 2019). The scope of that study was to collect samples from ponds at five SSOM facilities and two yard waste facilities over three sampling events. The study concluded that PFAS are prevalent in compost contact water at concentrations higher than in Minnesota's ambient groundwater.

The types of PFAS most relevant in compost contact water are terminal PFCAs, such as PFHxA, and PFSAs, such as PFOS, associated with composted items (Choi et al. 2019; Wood Environment & Infrastructure Solutions Inc. 2019). The selection of target PFAS for consideration in this study was detailed in Section 2.3, and concentrations of the target PFAS assumed to be present in compost contact water are summarized in Table 9-1. These values were developed to support preliminary design and cost estimating and are not intended to reflect the full range of PFAS concentrations potentially present in compost contact water.

Table 9-1 Assumed influent concentrations and treatment goals for target PFAS for compost contact water (all units in ng/L)

PFAS	Unit	Typical Concentrations[1]	High Concentrations[1]	Treatment Target
PFBA	ng/L	450	1,500	5
PFBS	ng/L	10	25	5
PFHxA	ng/L	500	2,000	5
PFHxS	ng/L	10	100	5
PFOA	ng/L	30	100	5
PFOS	ng/L	20	1,500	5

[1] Data summarized from Wood Environment & Infrastructure Solutions Inc. (2019).

9.1.3 Water Quality Assumptions for Pretreated Compost Contact Water

Compost contact water quality can vary significantly depending on specific feedstocks, the phase of composting, and specific operations of individual facilities.

Compost contact water typically has high nitrogen concentrations and high organic content with relatively low bioavailability (BOD_5:COD ratios are on the order of 0.1 to 0.5) because readily biodegradable organics are largely removed in the composting process. Other potential contaminants associated with compost contact water include heavy metals, suspended solids, chloride and other dissolved solids, and emerging organic contaminants, such as plasticizers (Roy et al. 2018).

Non-PFAS water quality parameters have significant implications for PFAS treatment. The water quality assumed for this study as initial untreated and pretreated water ahead of PFAS treatment is shown in Table 9-2. A range of concentrations is shown, reflecting the expected variability in water quality. This study assumes most facilities accept both food waste and yard waste as feedstocks, although water quality represented in Table 9-2 is based on compost contact water from a variety of facilities. Alternatives that do not require pretreatment (Alternatives 8a and 8b) are assumed to accept water with concentrations typical of raw compost contact water. Water quality targets for PFAS separation technologies requiring pretreatment (Alternatives 1a and 5a) are also shown. They reflect assumed water quality entering the PFAS management process based on equipment fouling potential outlined in Section 5.2. One example of a pretreatment train and costs that may be able to provide this water quality at commercial composting facilities is discussed in Section 9.4.

Table 9-2 Assumed initial and pretreated compost contact water quality

Parameter	Units	Assumed Water Quality in Untreated Compost Contact Water			Assumed Concentration in Pretreated Water (Influent for all Other PFAS Management Alts Evaluated)
		Typical Value (Assumed Influent Water Quality for Alts 8a and 8b)	Minimum Value	Maximum Value	
pH[1]	s.u.	7.5	4.0	8.0	7.5
TDS[2]	mg/L	6,000	Not reported	Not reported	6,000
TSS[1]	mg/L	1,000	10	34,000	1[3]
COD[1,4]	mg/L	25,000	400	185,000	90[5]
BOD_5[1,4]	mg/L	6,000	20	90,000	60[3]
TOC[4]	mg/L	750	<50	18,000	20[3]
Conductivity[1,4]	mS/cm	20	1.0	80	20
Hardness, as $CaCO_3$[4]	mg/L	1,800	1,100	2,400	1,800
Alkalinity[2]	mg/L as $CaCO_3$	1,800	Not reported	Not reported	1,800
Chloride[1]	mg/L	2,000	30	8,700	2,000
Sulfate	mg/L	Not reported	Not reported	Not reported	--
Sodium	mg/L	Not reported	Not reported	Not reported	--
Potassium	mg/L	Not reported	Not reported	Not reported	--
Calcium[4]	mg/L	360	280	460	360
Magnesium[4]	mg/L	210	100	300	210
Ammonia-N[1]	mg/L	800	5	2,300	800[6]
Phosphorus[1]	mg/L	30	0.5	500	30[6]
Iron[3]	mg/L	30	20	50	0.5[3]
Manganese	mg/L	Not reported	Not reported	Not reported	0.5[3]

[1] Concentrations shown are based on values in Tables 1, 2, and 3 of Roy et al. (2018).

[2] A typical value could not be established for TDS or alkalinity from referenced literature. The values shown were calculated based on the typical concentrations shown for the other constituents. The value shown for total dissolved solids only includes inorganic constituents; thus, it likely represents a low estimate.

[3] Based on the pretreatment discussion in Section 5.2.

[4] Concentrations shown are based on values provided in Table 7 of Krogmann and Woyczechowski (2000). Hardness was calculated based on reported concentrations of calcium and magnesium.

[5] The concentration of COD is assumed to decrease based on removal of BOD_5 during biological pretreatment (e.g., MBR). It is assumed that $COD=1.5\times BOD_5$.

[6] Concentrations of ammonia and phosphorus are both expected to decrease through biological pretreatment (e.g., MBR), but the level of removal is uncertain.

9.2 PFAS Management Alternatives Evaluation Results

Each PFAS management alternative identified for water-phase PFAS separation and destruction in Section 4.1 was scored for each evaluation criterion described in Section 5.1 for compost contact water.

9.2.1 Evaluation Scoring Results

Alternatives evaluated for compost contact water, as described in Section 4.1, were as follows:

- Alternative 1a: GAC sorption followed by off-site, high-temperature incineration of GAC
- Alternative 5a: modified clay media sorption followed by off-site, high-temperature incineration of modified clay
- Alternative 8a: Foam fractionation followed by off-site, high-temperature incineration of foamate
- Alternative 8b: Foam fractionation followed by SCWO of foamate

Evaluation results are summarized in Table 9-3 and described in subsequent sections. Specific criteria and subcriteria are defined in Section 5.1. Based on the screening, Alternatives 1a, 5a, and 8a were retained for preliminary design and cost curve development.

Table 9-3 Alternatives evaluation results for compost contact water

Alternative Number		1a	5a	8a	8b
PFAS separation technology		**GAC**	**MC**	**FF**	**FF**
PFAS destruction technology		**Inc**	**Inc**	**Inc**	**SCWO**
Technical Feasibility	**weight**	***59***	***56***	***47***	***40***
PFAS separation efficiency	5	3	3	2	2
PFAS destruction efficiency	5	3	3	3	3
Degree of commercialization	3	3	2	2	1
Reliability of performance	2	1	1	2	2
Simplicity of operation/maintenance	4	3	3	2	1
Operator and public health	2	3	3	2	2
Economic Feasibility	**weight**	***44***	***39***	***40***	***34***
Relative capital costs	4	3	3	2	1
Relative operation and maintenance costs	3	3	3	2	3
Relative energy consumption	2	2	2	2	2
Relative complexity and cost of pretreatment	4	1	1	3	3
Applicability at scale	5	3	2	2	1
Byproducts Management	**weight**	**3**	**3**	**3**	**5**
Beneficial reuse opportunity for water or byproducts	1	1	1	1	1
Potential for media shifting of PFAS	2	1	1	1	2
Total Score		**106**	**98**	**90**	**79**
Retained?		**x**	**x**	**x**	

GAC = granular activated carbon, MC = modified clay, Inc = high-temperature incineration, SCWO = supercritical water oxidation, FF = foam fractionation

9.2.2 Technical Feasibility

GAC and modified clay media scored highest for **PFAS separation efficiency** relative to foam fractionation due to the limited ability of foam fractionation to separate short-chain PFAS into foamate. Both GAC and modified clay are adsorption processes with effluent water quality dependent on media breakthrough status for specific PFAS. The score of 3 assigned here reflects the assumption that breakthrough could be detected in lead vessel effluent and media changed out before PFAS reports to the lag vessel effluent.

All alternatives received a score of 3 for **PFAS destruction efficiency** because both high-temperature incineration and SCWO remove 99.9% or more of PFAS. However, both have the potential for additional PFAS formation (Krause et al. 2022; L. Winchell et al. 2022). Additional research is needed for both these technologies to demonstrate the degree of mineralization achievable and the nature of byproducts formed.

GAC media with high-temperature incineration received a score of 3 for **degree of commercialization** because these technologies have been widely applied in multiple industries for PFAS removal. Alternatives including modified clay media and foam fractionation received a score of 2 because they are commercially applied but in fewer industries. The alternative including SCWO received a score of 1 because the technology is still developing for PFAS removal, especially in the water phase. It is currently limited to pilot-scale installations.

Sorption media received a score of 1 for **reliability of performance** because flow rate variation can adversely affect the EBCT and associated mass transfer zone in sorption vessels, which can cause premature breakthrough. In addition, rapid breakthrough of short-chain PFAS could result in detectable concentrations in treated effluent, depending on monitoring frequency and mobilization time for changeout. Foam fractionation with high-temperature incineration and with SCWO both received a score of 2 because treatment efficiency is expected to be unaffected by changing water quality, flows, or environmental conditions, but some uncertainty remains due to limited commercial application.

Alternatives with sorption media (GAC and modified clay) combined with high-temperature incineration received the highest score for **simplicity of operation**. These alternatives would have one on-site treatment process for PFAS and low operational complexity, except for sorption media changeout events. Foam fractionation with high-temperature incineration received a score of 2 because it is expected to be more complex to operate than single-use sorption media and only uses one on-site process. Foam fractionation combined with SCWO received a score of 1 due to the complexity of operating both technologies on-site.

Sorption media (GAC and modified clay) alternatives with high-temperature incineration received a score of 3 for **operator and public health** because there are minimal risks associated with these types of treatment and destruction technologies. All other alternatives received a score of 2 because there is an increased risk to operators and public health due to one or more of the following: the addition of chemicals, high operating temperatures, or high operating pressures.

9.2.3 Economic Feasibility

The alternative with the highest **relative capital cost** is foam fractionation with on-site SCWO destruction because it requires two on-site processes and therefore received a score of 1. Foam fractionation with off-site, high-temperature incineration received a score of 2 due to lower on-site capital expenses. Media sorption with off-site, high-temperature incineration scored a 3 because the vessels and ancillary equipment needed are relatively inexpensive.

When considering **relative O&M costs (without energy)**, alternatives with single-use media disposed of via high-temperature incineration received a score of 3, with primary O&M costs associated with transport and off-site incineration of a small volume of sorption media. Foam fractionation with high-temperature incineration scored a 2 due to the aeration costs of foam fractionation and the cost of off-site, high-temperature incineration of foamate. Foam fractionation with SCWO treatment of the foamate received a score of 3 because it does not incur fees associated with off-site, high-temperature incineration.

Alternatives with high-temperature incineration or sorption media or foam fractionation foamate received a score of 2 for **relative energy consumption** because they require thermal incineration of waste materials. Foam fractionation followed by SCWO also received a score of 2, assuming that the energy required for SCWO treatment of foamate is similar to that required for high-temperature incineration of the same volume. SCWO of foamate is expected to require a co-fuel, such as diesel, to be self-sustaining. However, an on-site material, such as reject material or residuals, could potentially be used as a co-fuel source to reduce the purchase of external fuels.

GAC and modified clay sorption scored a 1 for **relative complexity and cost of pretreatment** because it would require pretreatment, as described in Section 9.5. Alternatives with foam fractionation scored a 3 because they do not require complex pretreatment.

GAC sorption with high-temperature incineration received a score of 3 for **applicability at scale**. Modified clay sorption and foam fractionation with high-temperature incineration each received a score of 2 because applications using these technologies are more limited. Foam fractionation with SCWO received a score of 1 because it includes two technologies with limited applications at this scale.

9.2.4 Byproducts Management

All four alternatives received a score of 1 for **beneficial reuse** because the water or solid byproducts produced are not expected to have valuable reuse opportunities.

All alternatives with high-temperature incineration received a score of 1 for **potential media shifting of PFAS.** This is due to the current uncertainty and ongoing industry research to investigate PFAS transformations and emissions associated with high-temperature incineration. Alternatives with SCWO destruction received a score of 2 because there are still unknowns regarding the mass balance of fluorine through SCWO reactors.

9.2.5 PFAS Management Alternatives for Preliminary Design

Based on the screening described in Section 9.2, three PFAS management alternatives were carried forward to preliminary design and cost analysis for compost contact water:

- Alternative 1a: GAC sorption followed by off-site, high-temperature incineration of GAC
- Alternative 5a: modified clay media sorption followed by off-site, high-temperature incineration of modified clay
- Alternative 8a: foam fractionation followed by off-site, high-temperature incineration of foamate

9.3 Alternative Preliminary Design and Cost Curve Development

9.3.1 Alternative 1a (Compost Contact Water)

For Alternative 1a, pretreated compost contact water meeting the water quality listed in Table 9-2 flows directly to lead-lag GAC adsorption pressure vessels. Once GAC media is exhausted with respect to the target PFAS (limited by PFAS breakthrough in this case), it would be sent to high-temperature incineration for destruction. This process flow is summarized in Figure 9-1.

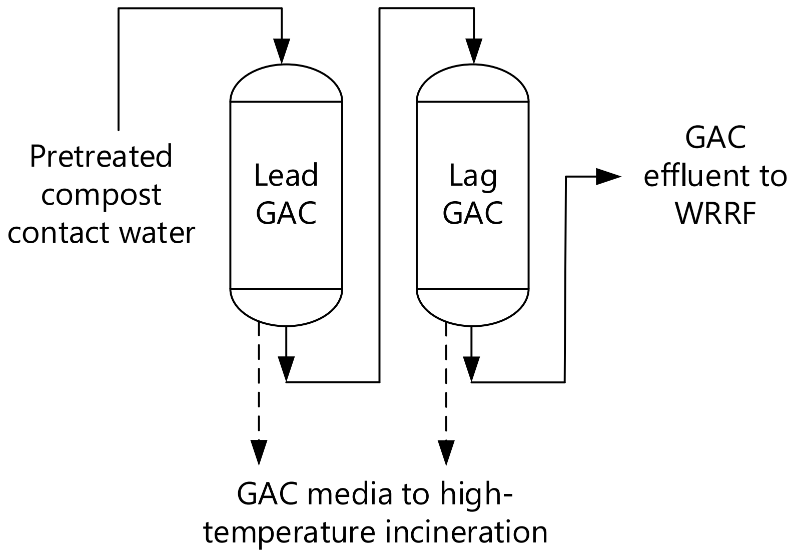


Figure 9-1 **Conceptual process flow diagram for Alternative 1a for compost contact water**

Key design basis assumptions for Alternative 1a are listed in Table 9-4, with additional details in Appendix E.

Table 9-4 Summary of design basis assumptions for Alternative 1a for compost contact water

Design Parameter	Basis		
	0.0014 MGD/ 1 gpm	0.014 MGD/ 10 gpm	0.14 MGD/ 100 gpm
Vessel capacity (lb)	90	750	10,000
Number of trains	2	1	1
Number of vessels per train	4	2	2
EBCT per vessel (min)	15	15	15
HLR (gpm/sq. ft)	0.5	3.2	2.0
Estimated bed volumes to breakthrough[1]	6,400	6,400	6,400
GAC disposal route	High-temperature incineration	High-temperature incineration	High-temperature incineration

[1] Defined as the estimated volume of water treated through the lead media vessel until the first detection of PFAS at 5 ng/L. Described further in Section 5.3.4 and Appendix D.

Figure 9-2 and Figure 9-3 show the estimated capital and annual O&M costs for the range of influent flow rates considered for compost contact water. These costs do not include the pretreatment costs summarized in Section 9.5. Detailed tables with supporting data for the cost curves are included Appendix E.

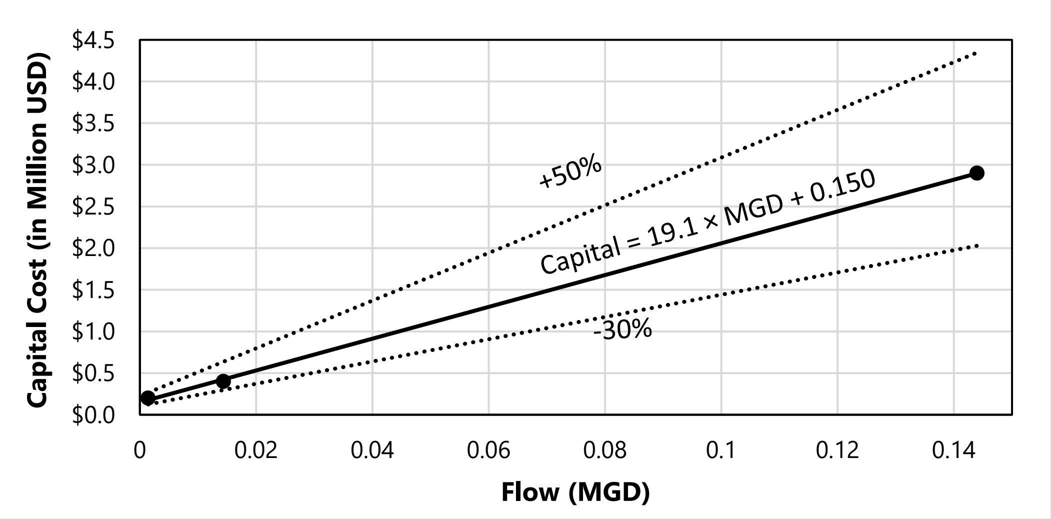


Figure 9-2 Capital cost curve for Alternative 1a for compost contact water

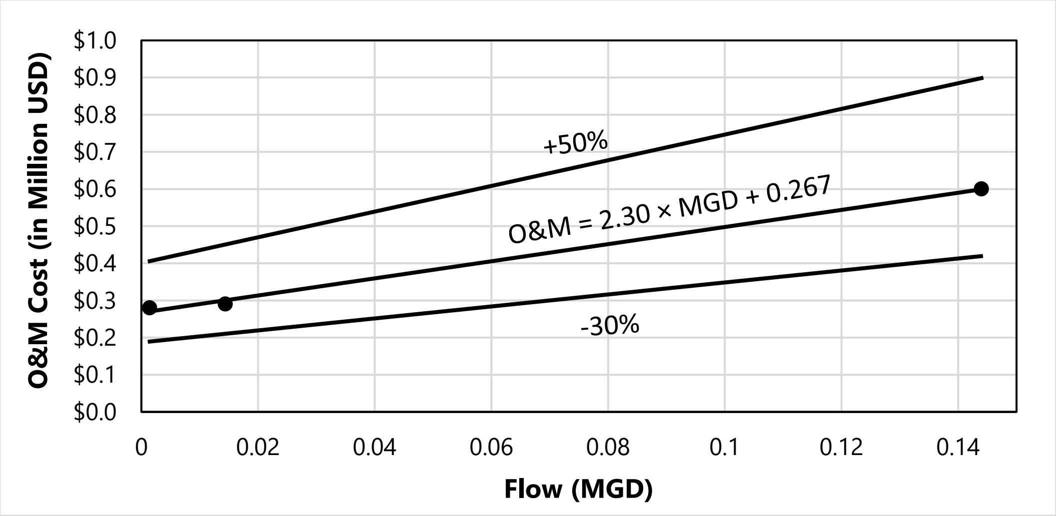


Figure 9-3 **O&M cost curve for Alternative 1a for compost contact water**

Estimated annual O&M costs are approximately 40% over the estimated capital costs at the lowest flow rate. Estimated annual O&M costs are similar to estimated capital costs at 10 gpm. The highest operating costs are associated with operation and maintenance labor and sorption media replacement.

Primary uncertainties for Alternative 1a for compost contact water include the following:

- Actual GAC breakthrough timing and requirements, depending on actual pretreated water quality
- Potential for GAC fouling limiting bed life over PFAS breakthrough
- Pretreatment processes needed and ability to meet pretreatment targets
- Location and incineration fees of a selected high-temperature incineration facility
- The capital costs for the lowest flow rates may change depending on facility preference for operational flexibility. The vessels included in this analysis are relatively small such that a typical valve tree included with larger vessels is not standard and was not included.

9.3.2 Alternative 5a (Compost Contact Water)

For Alternative 5a, pretreated compost contact water meeting the water quality listed in Table 9-2 flows directly to lead-lag modified clay adsorption pressure vessels. Once modified clay media is exhausted with respect to the target PFAS (limited by breakthrough of short-chain PFAS), the spent sorption media would be sent to high-temperature incineration for destruction. This process flow is summarized in Figure 9-4.

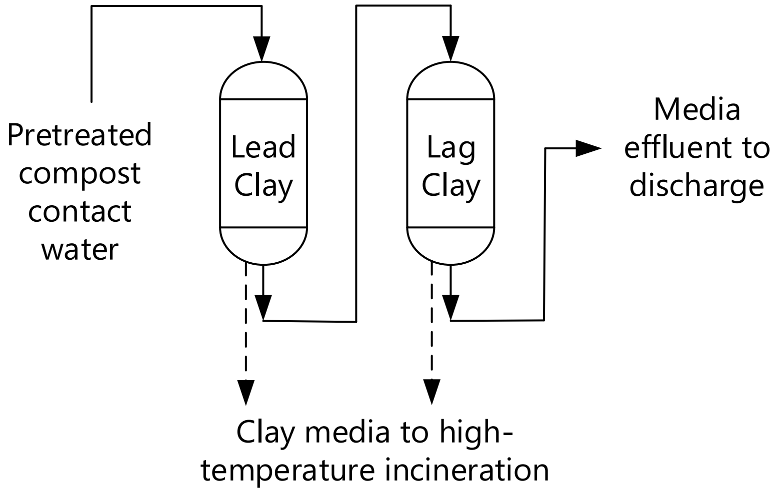


Figure 9-4 **Conceptual process flow diagram for Alternative 5a for compost contact water**

Key design basis assumptions for Alternative 5a are listed in Table 9-5, with a detailed design basis in Appendix E.

Table 9-5 **Summary of design basis assumptions for Alternative 5a for compost contact water**

Design Parameter	Basis		
	0.0014 MGD/1 gpm	0.014 MGD/10 gpm	0.14 MGD/100 gpm
Vessel capacity (cu. ft)	3	5	40
Number of trains	1	3	4
Number of vessels	2	6	8
EBCT per vessel (min)	10	10	10
HLR (gpm/sq. ft)	0.9	1.9	3.5
Estimated bed volumes to breakthrough[1]	10,000	10,000	10,000
Media disposal route	High-temperature incineration	High-temperature incineration	High-temperature incineration

[1] Defined as the estimated volume of water treated through the lead media vessel until the first detection of PFAS at 5 ng/L. Described further in Section 5.3.4 and Appendix D.

Figure 9-5 and Figure 9-6 illustrate estimated capital and annual O&M costs for a range of influent flow rates. These costs do not include pretreatment costs described in Section 9.5. Detailed capital and O&M cost estimates are in Appendix E.

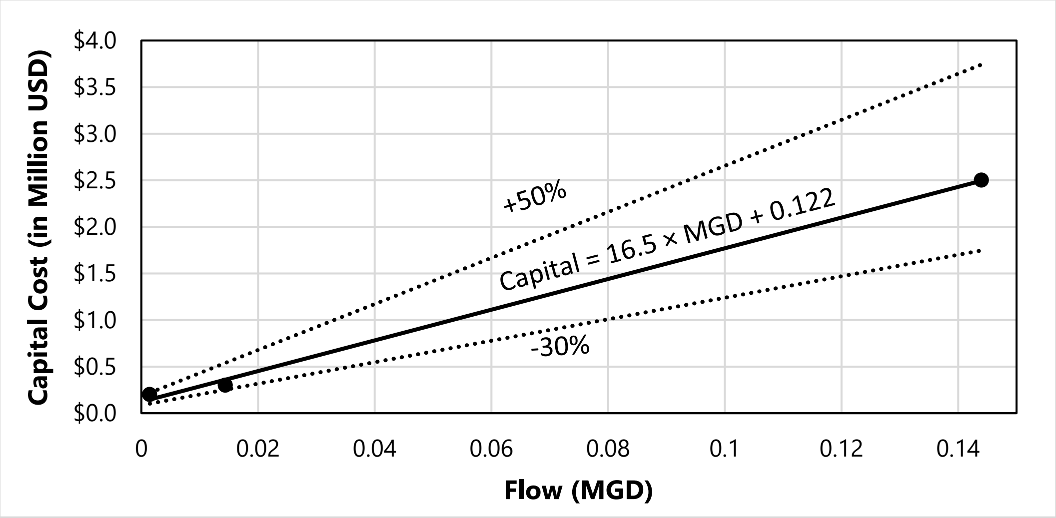


Figure 9-5 Capital cost curve for Alternative 5a for compost contact water

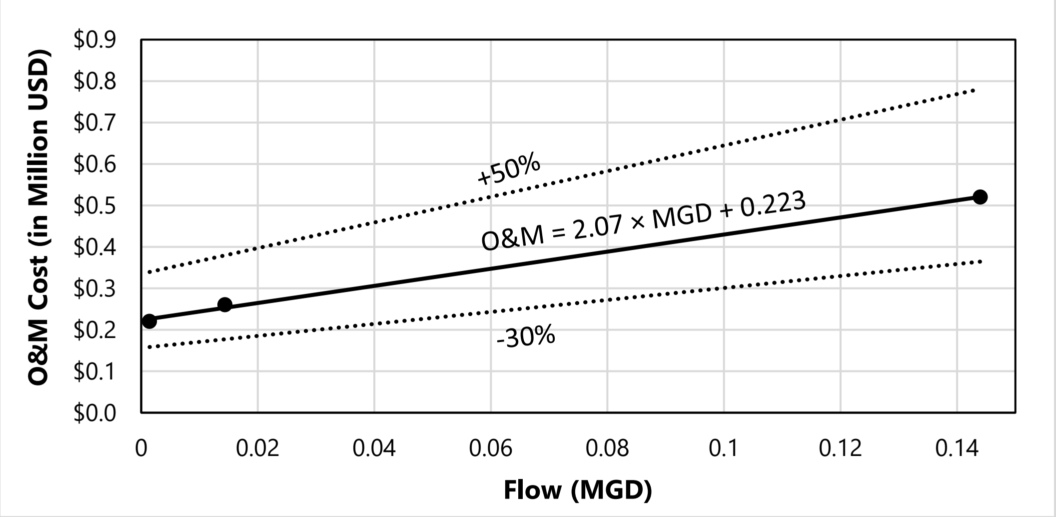


Figure 9-6 O&M cost curve for Alternative 5a for compost contact water

Estimated annual O&M costs and estimated capital costs for Alternative 5a are similar to estimated costs for Alternative 1a. The highest operating costs are associated with labor and sorption media replacement and incineration.

Primary uncertainties for Alternative 5a for compost contact water treatment include the following:

- Actual PFAS breakthrough timing through modified clay media, depending on media performance and actual pretreated water quality
- Potential for modified clay fouling to limit bed life over PFAS breakthrough
- Pretreatment processes needed and ability to meet pretreatment targets
- Location and incineration fees of a selected high-temperature incineration facility

- The capital costs for the lowest flow rates may change depending on facility preference for operational flexibility. The vessels in this analysis are relatively small such that a typical valve tree included with larger vessels is not standard and was not included.

9.3.3 Alternative 8a (Compost Contact Water)

For Alternative 8a, compost contact water would be pumped to a series of foam fractionation vessels. The contact water initially goes through a bag filter to remove suspended solids, avoiding solids buildup in the foam fractionation vessels. The foam generated from the primary foam fractionation vessel is further enriched in a secondary foam fractionation vessel. The air that generates the foam in the closed foam fractionation vessel is discharged to carbon canisters to capture volatile organics. The resulting foam is collected and transported to a high-temperature incineration facility for destruction. This process flow is summarized in Figure 9-7.

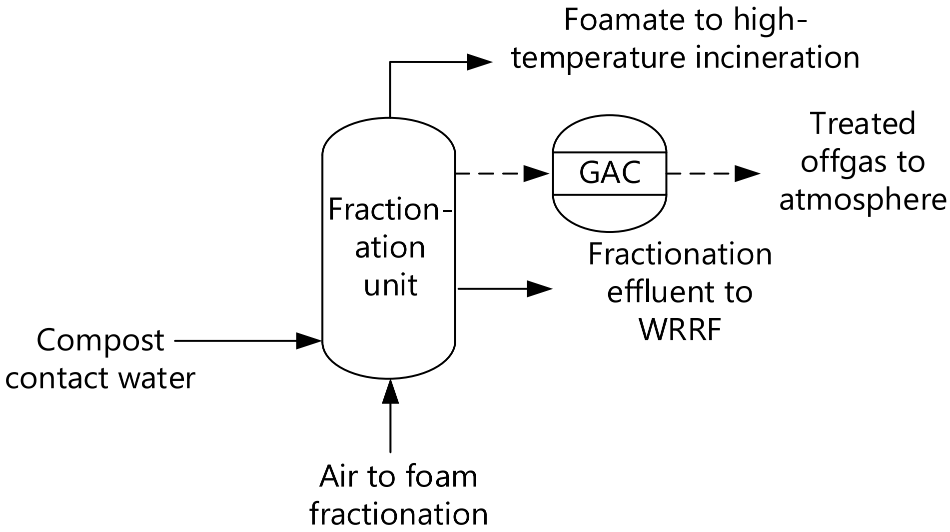


Figure 9-7 **Conceptual process flow diagram for alternative 8a for compost contact water**

Key design assumptions for Alternative 8a are listed in Table 9-6, with an additional detailed design basis in Appendix E.

Table 9-6 **Summary of design basis assumptions for Alternative 8a for compost contact water**

Design Parameter	Basis		
	0.0014 MGD/1 gpm	0.014 MGD/10 gpm	0.14 MGD/100 gpm
Number of primary vessels	2	2	4
Number of secondary vessels	1	1	2
Assumed concentration factor[1]	1,000,000	1,000,000	1,000,000
Expected foamate flow rate (gpd)	0.0014	0.014	0.14

[1] Based on vendor input.

Figure 9-8 and Figure 9-9 show estimated capital and annual O&M costs, respectively, for the range of influent flow rates considered for compost contact water. Detailed tables with supporting data for the cost curves are in Appendix E.

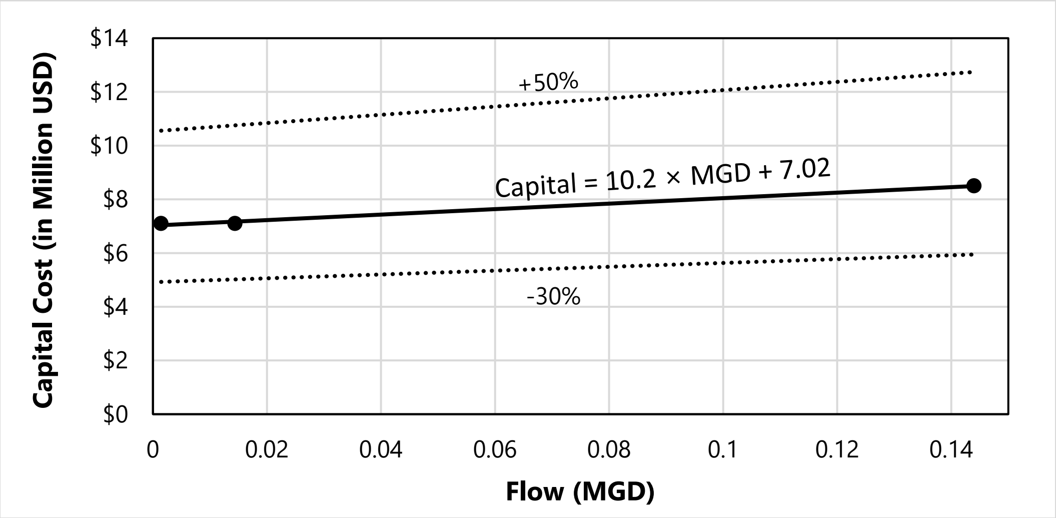


Figure 9-8 **Capital cost curve for Alternative 8a for compost contact water**

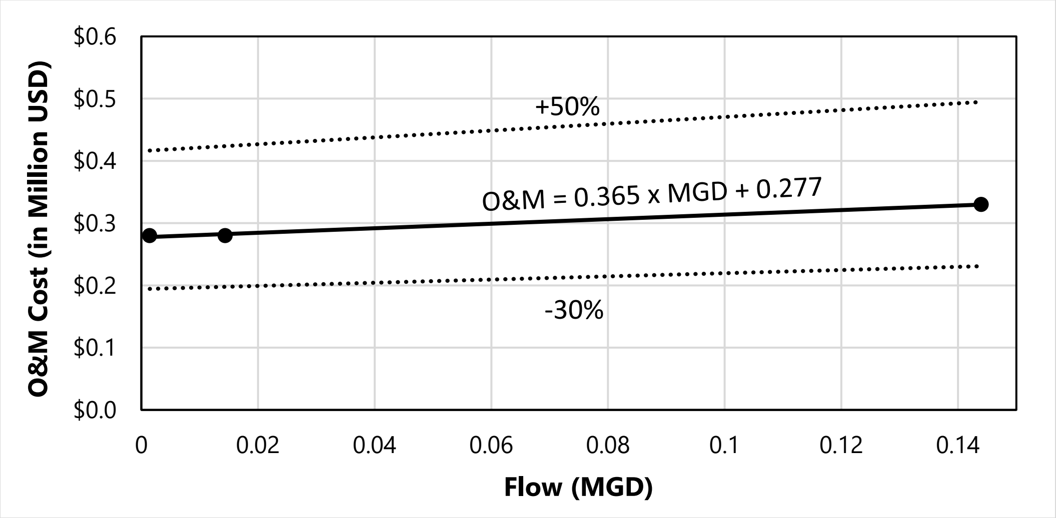


Figure 9-9 **O&M cost curve for Alternative 8a for compost contact water**

Estimated capital and O&M costs for compost contact water are relatively independent of flow rate due to the modular nature of foam fractionation equipment and current vendor offerings. Capital costs for foam fractionation are approximately 3–40 times higher than GAC adsorption (Alternative 1a), and O&M costs are comparable between these two alternatives. The highest operating costs are associated with utilities, maintenance, and labor.

Primary uncertainties for Alternative 8a for compost contact water include the following:

- Actual concentrating factor of the foam (assumed to be 1,000,000)

- The removal efficiency of short-chain PFAS (such as PFBA) and the potential usage rate of cationic surfactants or enhancements to improve the removal efficiency. Polishing treatment (such as media sorption) on the treated water stream from foam fractionation may be needed to achieve the PFAS performance targets of this study. This would increase the capital and O&M cost estimates provided above.
- Location and incineration fees of a selected high-temperature incineration facility

9.4 PFAS Removal Performance and Reliability

The relative ability of each alternative to meet treatment goals of <5 ng/L for target PFAS is summarized in Table 9-7. Alternatives 1a and 5a are expected to meet the PFAS treatment targets most of the time, with some risk of rapid breakthrough of short-chain PFAS and frequent media changeout required. Currently, it is uncertain if foam fractionation (Alternative 8a) can remove 90% of short-chain PFAS. If not, this would result in pass-through of PFAS such as PFBA, PFBS, PFHxA, and possibly others into the treated water phase at concentrations higher than the 5 ng/L target. Specific references for PFAS removal efficiency are detailed further in Appendix C.

Table 9-7 PFAS removal performance and reliability for compost contact water treatment alternatives

Target PFAS	Assumed Influent Concentration (ng/L)	Percent Removal to Achieve <5 ng/L	Relative Ability to Consistently Meet Targets		
			1a: GAC	5a: Modified Clay	8a: Foam Fractionation with Incineration
PFBA	450	99%	Moderate	Moderate	Low
PFBS	10	50%	Moderate	Moderate	Low
PFHxA	500	99%	Moderate	Moderate	Moderate
PFHxS	10	50%	High	High	High
PFOA	30	83%	High	High	High
PFOS	20	75%	High	High	High

9.5 Example Pretreatment for PFAS Separation Technologies Applied to Compost Contact Water

This section summarizes the conceptual design and capital costs for the pretreatment of compost contact water to limit the fouling of pressure vessel media sorption for PFAS treatment (Alternatives 1a and 5a). Other alternatives considered applicable for compost contact water (Alternative 8a and 8b—foam fractionation) are not expected to require pretreatment aside from filtration for TSS reduction. Currently, no commercial composting facilities in Minnesota have active treatment for contact water. Thus, implementing PFAS water treatment at composting facilities would require significant site upgrades and equipment installation.

Based on the assumed water quality in Table 9-2, contact water would need to be treated for TSS, TOC, and iron removal ahead of sorption media pressure vessels (Alternatives 1a and 5a) for PFAS treatment.

The conceptual pretreatment system for compost contact water is assumed to include the following:

- Aeration—to oxidize iron
- Chemical precipitation and sedimentation—to remove oxidized iron, a fraction of the TOC, and fine suspended solids
- Acid addition—to lower pH and prevent unionized ammonia toxicity during biological treatment (discussed in detail in Section 8.5)
- MBR—to remove BOD_5, TOC, and residual suspended solids

The conceptual pretreatment process flow is shown in Figure 9-10.

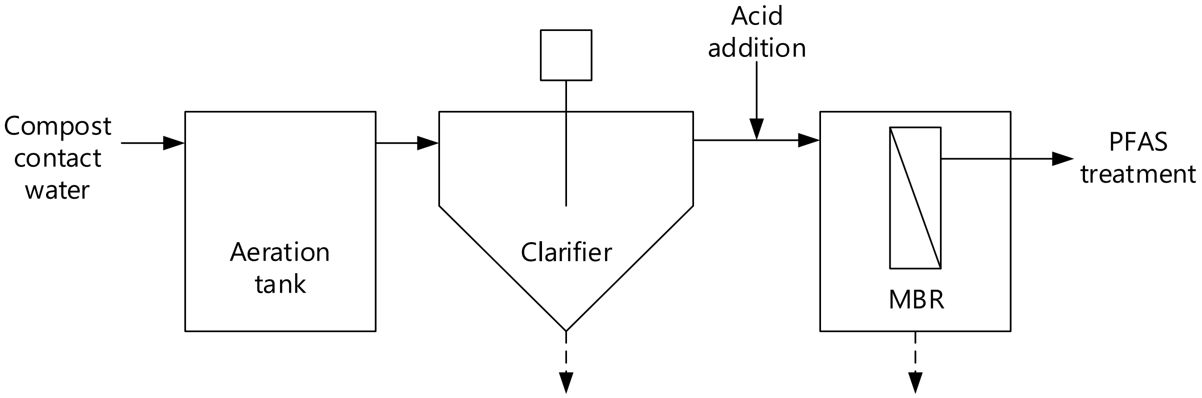
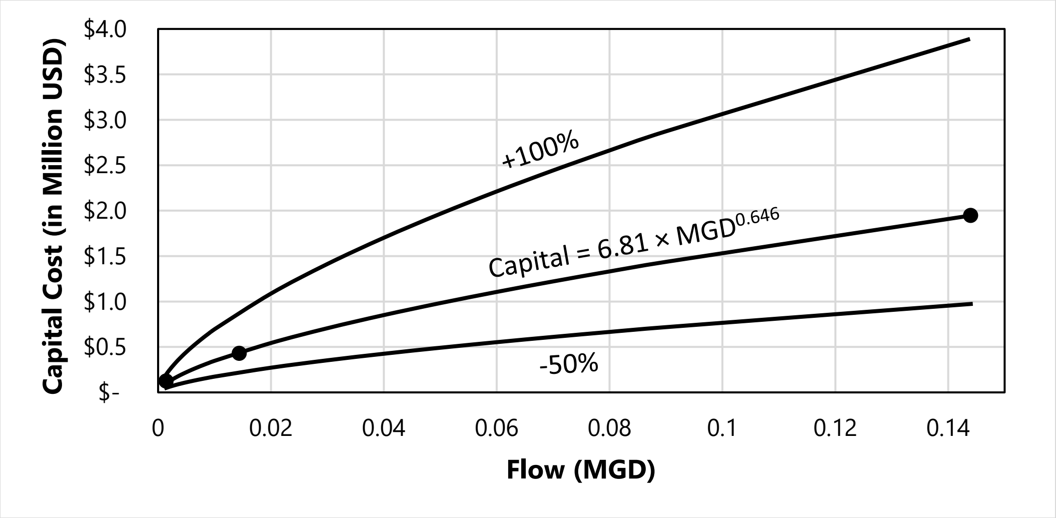


Figure 9-10 Conceptual pretreatment process flow diagram for compost contact water

MBR systems require consistent flows to maintain healthy biomass. Because compost contact water flows are variable and dependent on precipitation, the treatment system will need sufficient flow equalization to maintain continuous flow during dry periods (e.g., summer and winter). It is assumed that existing contact water collection ponds could be used for flow equalization, with minor modifications as necessary. Thus, construction costs for equalization basins are not included in the capital costs shown below. If sufficient flow equalization is not practical for a specific facility, alternative pretreatment other than MBR could be evaluated to allow pretreatment on an as-needed basis (i.e., batch-wise) instead of MBR pretreatment, which requires continuous flow. For example, greensand filtration could be used for iron and TSS removal with GAC adsorption for TOC removal. An individual, site-specific feasibility study should be completed to evaluate the appropriateness of the conceptual pretreatment presented here, or alternative pretreatment strategies should be identified.

High salts and metals, low biodegradability, and variable flow rates make compost contact water challenging to treat using conventional treatment methods. Depending on water quality and site-specific conditions, pretreatment using an MBR followed by GAC adsorption for PFAS removal may not be effective. First, the low biodegradability of the organic matter present means that the pretreatment target for TOC may not be met. This could result in high concentrations of TOC in sorption vessel influent, using up sorption sites and requiring significantly faster changeout frequencies than anticipated to maintain effluent PFAS concentrations below detection limits. Second, iron may not be removed to the pretreatment targets through coagulation and settling and could precipitate on the sorption media, ultimately blocking pore spaces and requiring backwashing or frequent media changeouts.

Capital cost estimates for this pretreatment train are shown in Figure 9-11. Costs are Class 5 (AACE) cost estimates with an accuracy range of +100%/-50%. This cost curve should only be used for the flow range shown (1–100 gpm; 0.0014–0.14 MGD).



MBR costs from Lo, McAdam, and Judd (2015). Other equipment costs estimated from CapdetWorks (Hydromantis, v4.0). The solid line reflects the estimated cost curve. The dashed lines reflect +100%/-50% accuracy range. All costs scaled to November 2022 using ENR CCI (13175).

Figure 9-11 Capital cost curve for new MBR installations with pre-aeration, sedimentation, and chemical addition for compost contact water pretreatment

The conceptual pretreatment cost estimates did not include a detailed feasibility study or any design for site-specific requirements. Pretreatment needs are expected to vary by facility due to differences in existing infrastructure, water quality, space availability, and other site-specific constraints. Thus, the degree of detail included in this Report for pretreatment is less than that provided for PFAS treatment alternatives in subsequent sections. The conceptual pretreatment costs shown are intended to provide an order-of-magnitude estimate for potential pretreatment costs. **Facilities looking to implement PFAS treatment should conduct site-specific treatment evaluations prior to capital planning or implementation.**

9.6 Other Considerations for Compost Contact Water

The PFAS management alternative most appropriate for a given site will depend on space constraints, initial water quality, operational and treatment targets, and permit conditions. Sites targeting only long-chain PFAS will likely find foam fractionation (as included in Alternative 8a) the most cost-effective separation technology because it does not require pretreatment. However, foam fractionation's efficacy for separating short-chain compounds remains limited at this time. Modified clay (Alternative 5a) is still being vetted as a sorption media for PFAS separation but may have specific advantages over GAC (Alternative 1a), such as lower capital cost and footprint.

The decision to select one of these alternatives and implement them on-site will vary site by site and be based on the PFAS management strategy of the WRRF receiving the discharge. PFAS management strategies should consider if an on-site treatment is necessary. For example, if a downstream WRRF is implementing PFAS treatment or is willing to accept compost contact water as is, PFAS management may not be needed at the composting site. Other considerations for removing and destroying PFAS from compost contact water include the following:

- Commercial SSOM composting facilities offer a critical means to reduce organic waste going to landfills and produce a saleable product (Minnesota Pollution Control Agency 2022). However, compost facilities have a very low-cost margin and limited ability to implement costly treatment of PFAS in contact water. Any level of on-site treatment of PFAS in contact water will likely require increasing the costs to dispose of organic materials at composing facilities or require external funding to keep these facilities operational.
- While this Report identifies potential options for active treatment of PFAS in compost contact water, reducing PFAS loads in compost feedstocks in parallel with active water treatment would be a complementary way to reduce PFAS in compost contact water. Possible ways to reduce PFAS in compost feedstocks include (U.S. EPA 2021c):
    - Feedstock restrictions—identify PFAS loads in compost feedstock materials (e.g., food contact materials) and implement outreach programs to limit the input of these materials.
    - Phase-outs and bans on the use of fluorinated chemicals in compostable food materials, for example, refer to the standard for fluorinated chemicals from the Biodegradable Products Institute (Biodegradable Products Institute 2020).
- Reducing the volume of contact water is another strategy that could be implemented in conjunction with active water treatment to reduce the overall size of the treatment system. Volume reduction could be accomplished by converting an open compost facility to an enclosed facility to limit water contributions from precipitation. A site-specific feasibility study would likely be needed to evaluate the costs and benefits of operating an enclosed composting facility versus an open composting facility and the expected impact on treatment system sizing.
- High salt and metal concentrations, expected low biodegradability of organics, and variable flow rates of compost contact water make it challenging to treat using conventional treatment methods like GAC adsorption for PFAS removal. Future management of PFAS at composting facilities should target low-energy, low-complexity treatment that can handle variable flows, variable water qualities, and treatment options that can be operated as needed to manage periods of limited or no contact water generation.

- GAC treatment (Alternative 1a) was identified as the likely lowest-cost option, even when considering the inclusion of pretreatment unit operations. However, long-term GAC performance is uncertain, especially the feasibility of treating short-chain, terminal PFAS like PFBA with a relatively high load of background organics.

- Modified clay (Alternative 5a) has slightly higher capital costs than Alternative 1a. The benefit of Alternative 5a over Alternative 1a may be higher water throughput before breakthrough of target PFAS. However, on-site testing would be required to determine the most effective sorption media.

- Foam fractionation (Alternative 8a) with off-site disposal of the foamate is a promising alternative with similar estimated ongoing O&M costs as GAC but would likely require a higher upfront capital cost. Foam fractionation would also likely require on-site feasibility testing to demonstrate whether it can meet PFAS treatment targets, especially for short-chain PFCAs.

- Other on-site destruction strategies for PFAS management will likely become more cost-effective as these technologies are further vetted and implemented. Future technology development for PFAS treatment is discussed further in Section 11.8.

10 Regional PFAS Management Facility Evaluation

Regional management of the waste streams containing PFAS described in the prior sections of this Report is considered here as a potential option to provide cost savings to multiple customers by centralizing PFAS destruction in one or several large, centralized locations rather than having on-site destruction technologies installed at individual sites. Here a "regional" facility is any facility that treats concentrated PFAS waste streams from multiple sites, as compared to the previous sections of this study, which addressed PFAS wastes from a single WRRF, landfill, or composting facility.

Existing WRRFs currently serve as regional collection facilities for PFAS-impacted municipal and industrial streams, as well as direct-piped or hauled landfill leachate and compost contact water, However, these facilities were not designed to remove or destroy PFAS. If PFAS treatment needs to be implemented at WRRFs, they could also serve as regional facilities for PFAS management and destruction, provided they have available space for expansion.

10.1 Products for Regional Facility PFAS Destruction by Waste Stream

The products that could potentially be managed at a regional facility include solids or concentrated liquids. From WRRFs, this would include spent sorption media or concentrated liquids such as foam fractionation foamate or AIX regeneration still bottoms. Biosolids would likely be dewatered prior to management at a regional facility. Landfill leachate and compost contact water could potentially be sent to a regional facility without concentration, depending on the flow requiring treatment. Some of these facilities effectively practice regional management already by sending or trucking their effluents to a central WRRF. More practically, spent sorption media or concentrated liquids could be sent to a regional PFAS destruction facility.

10.2 Technologies for Regional Facility PFAS Destruction

The commercially available technologies known to destroy PFAS (high-temperature incineration, GAC reactivation, pyrolysis/gasification with thermal oxidation, and SCWO) are much more complex than typical waste management techniques in the state of Minnesota, requiring specialized training for operators. A regional PFAS residuals management facility could lower the number of PFAS destruction systems for biosolids and the number of specialized operators required within a certain geographic area.

PFAS destruction facilities currently at scale are all high-temperature processes located outside the State of Minnesota. Potential economies of scale may be realized for these facilities due to their high capital costs as well as the associated potential for heat retention and energy recovery inherent in larger facilities. Regional savings would also be realized by reduced transportation costs for destruction of these materials outside Minnesota. For this study, regional destruction resources are evaluated for the final destruction of PFAS-containing waste streams using off-site high-temperature incineration or GAC reactivation (Alternatives 1a, 2a, 2b, 5a, 6a, 6b, and 8a developed for design and cost estimating).

Other regionalization options that may become feasible in the future include regional disposal of smaller volumes of foam fractionation foamate or AIX regeneration still bottoms using emerging destruction technologies such as SCWO, HALT, or electrochemical oxidation.

10.3 Specific Considerations for Regional Facilities

Regional facilities can be economically feasible and would provide resilience to individual facilities as current hauling and disposal rates for PFAS-laden waste streams increase. Prioritizing proximity to individual facilities in selection regional facility locations to reduce material hauling costs are likely to be critical for financial viability.

The operation and management of regional facilities would be challenging and may require establishment of a specific entity/partnership to finance and operate the facility. Various management strategies and financial approaches can be developed. In one option (fee-based), costs incurred by the facility (capital, operation, and maintenance) could be recovered by tipping or disposal fees collected from individual facilities. This model may require higher fees and would not guarantee long-term collaboration with individual facilities. An alternative would separate the design and construction costs of the regional facilities in an up-front fee to WRRF partners (ownership). This would guarantee the participating individual facilities access to a portion of the regional facility capacity. The participating utilities typically contract with the regional facility owner to provide a portion of the funding for construction, maintenance, and facility operation. Contract terms vary widely but typically include fixed annual fees, periodic equipment replacement fees, and fees per unit of biosolids delivered to the regional facility. An example of this economic framework is detailed in Section 10.5.3.

10.3.1 Summary of Existing Regional Facility Networks

An existing network of high-temperature incineration facilities for hazardous waste currently accepts GAC and other sorption media containing PFAS. This network comprises about 10 facilities in the country, including those operated by Clean Harbors Environmental Services and Veolia North America, with none in Minnesota.

Regional GAC reactivation facilities are operated by GAC vendors, with approximately 17 facilities in the country (U.S. EPA 2020), with none in Minnesota. These facilities have processes for GAC receiving and reactivation, followed by thermal oxidation of released gasses, as described in Section 3.4.2.

10.3.2 Potential New Regional Facility Concepts

Four types of regional PFAS destruction facilities were considered for evaluation in this report:

- High-temperature incineration facility for sorption media, AIX resins, and other wastes,
- Pyrolysis/gasification facility with thermal oxidation for WRRF biosolids,
- AIX regeneration facility for spent, regenerable AIX resins, and

- SCWO facility accepting a variety of wastes, including biosolids, sorption media, AIX resin, and concentrated liquid waste streams.

Given that high-temperature incineration is currently the most widely used destruction alternative for PFAS, **potential cost scenarios and other considerations were developed for a hypothetical Minnesota high-temperature incineration facility.** A new regional high-temperature incineration facility could incinerate sorption media, such as GAC, AIX resin, or modified clay media at high temperatures above 1,000°C, as described in Section 3.4.1. This facility could also potentially accept other PFAS-containing wastes such as biosolids, RO concentrate, foam fractionation foamate, or AFFF, though these feeds are not considered in this cost evaluation for simplicity. Considerations and costs for a potential new regional high-temperature incineration facility are likely similar to a new regional GAC reactivation facility, as the processes and temperatures required are relatively similar.

A pyrolysis/gasification system with thermal oxidation was chosen for the development of cost scenarios and considerations for a regional PFAS destruction facility for biosolids. Pyrolysis/gasification systems with thermal oxidation are commercially available at full-scale. A regional pyrolysis/gasification facility would accept dewatered biosolids from municipal WRRFs rather than receiving liquid biosolids. Despite being dewatered, the biosolids would still contain more than 70% water, making the hauling distance to a regional facility and associated costs an important consideration. Receiving schedules and feed rates would need to be considered in planning a regional pyrolysis/gasification facility.

A regional AIX regeneration facility would use organic solvent and brine mixtures to regenerate AIX resin for reuse, as described in Section 3.3.4. This is not a destruction technology but produces a brine with high concentrations of PFAS. This technology has been implemented at selected sites with AIX resin use rates large enough to merit the capital expense of a regeneration facility. However, this option was not considered for a potential regional facility due to the limited application of AIX regeneration and the fact that it is not a destruction technology. This option could be considered in the future as the number of AIX regeneration applications continue to be developed.

A regional SCWO facility could potentially accept a combination of wastes, including biosolids, spent sorption media and AIX resins, and concentrated waste streams from liquid waste facilities. SCWO was previously detailed in Section 3.4.3. Specific wastes could include RO concentrate, foam fractionation foamate, and sorption media from WRRFs, landfills, and composting facilities. Planning and designing a multi-media SCWO facility would need to consider the range of waste mixtures that would make the facility economically viable. This option was not considered for detailed evaluation at this time because current SCWO applications are currently limited to small and pilot scale.

10.4 Sorption Media High-Temperature Incineration Facility Concept-Level Design

10.4.1 Design Basis and Equipment Needs

A new high-temperature incinerator targeted for sorption media is expected to require the following components:

- Rotary kiln (1,000°C)
- Secondary combustion chamber (1,100°C)
- Spray dryer
- Acid gas scrubbers to neutralize HF and other acids
- Particulate baghouse filtration
- Liquid scrubber to remove sulfur and NO_x
- Waste receiving equipment and storage tanks
- Ash treatment process and metal recovery process
- Residual solids load-off facility
- Wastewater neutralization and treatment
- Boilers to provide heat

Specific unit sizes and requirements were not estimated because costs are scaled from reported expenses for a similar facility. The conceptual high-temperature incineration process is illustrated in Figure 10-1. Mineralization of PFAS is expected to produce significant quantities of corrosive hydrofluoric acid, which requires high-level alloys to sustain long-term operation at the high operating temperatures required.

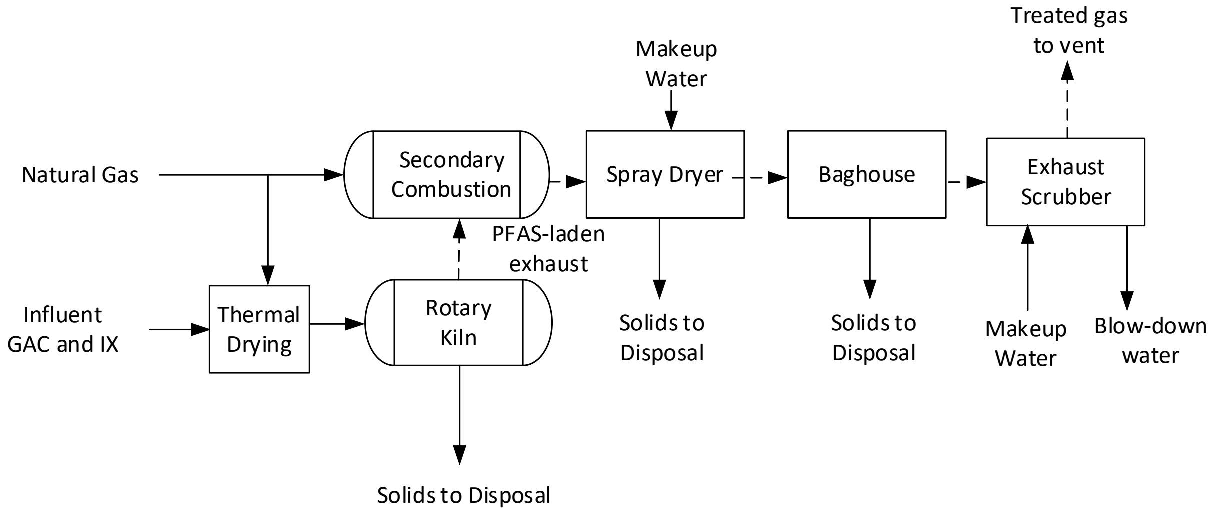


Figure 10-1 **Conceptual process flow diagram for sorption media regional high-temperature incineration facility**

10.4.2 Cost Estimates

Estimating the cost of a new high-temperature solid waste incinerator facility capable of effectively destroying fluorocarbon bonds (i.e., 1,100˚C) is difficult because the system requires highly specialized materials (e.g., combusting fluorocarbons generates hydrofluoric acid that will corrode conventional steel and refractory materials). It also requires ancillary operations to support raw waste handling/storage and waste ash processing. In 2016, Clean Harbors completed a $120 million expansion to add a third high-temperature hazardous waste incinerator to their El Dorado, Arkansas facility. The facility expansion can accept 60,000 tons/year of solid and liquid waste, including PFAS-containing wastes (Arkansas DEQ 2013). The Clean Harbors El Dorado Waste Incinerator Project's incinerator loading design basis (and the corresponding fluorocarbon loading) is not publicly available. However, for this report, the project was assumed to be a sufficient surrogate for estimating the costs of a new similarly sized high-temperature solid and liquid waste incinerator facility.

A new regional facility would also require the construction of several systems that were pre-existing and thus not included in the El Dorado expansion costs. These include waste receiving, ash treatment, scrubbers, and boilers. Therefore, the 2016 Clean Harbors El Dorado Incinerator Expansion Project cost underrepresents the costs for a new waste incinerator facility, and a scale factor of 1.5 (low-range) to 2.5 (high-range) was applied to the Clean Harbors El Dorado Waste Incinerator Project's cost. In addition, costs were adjusted for inflation using the *Chemical Engineering* magazine's Chemical Engineering Plant Cost Index (CEPCI) (Lozowski n.d.).

Table 10-1 presents the estimated 2022 cost for constructing a new, greenfield high-temperature waste incinerator facility capable of processing 60,000 tons/year of PFAS-containing waste streams to match the size of the facility expansion discussed above. This capacity is similar to the rate of spent GAC from 350 MGD of WRRF effluent treatment for PFAS using the media use rates estimated for this study. This cost estimate does not include costs associated with engineering or permitting, which could be substantial, depending on the degree of public support and comment.

Table 10-1 Estimated capital costs for a regional high-temperature incineration facility in Minnesota

Variable	Value
2016 Clean Harbors El Dorado Waste Incinerator expansion cost	$120,000,000
Inflation adjustment factor (CEPCI May 2022/2016)	53.4% (831.1/541.7)
Estimated ancillary plant infrastructure scale factor[1]	2.0
Estimated 2022 new greenfield waste incinerator facility cost range (2022 dollars)	$370,000,000 Range: (-30%) $260,000,000 to (+50%) $550,000,000

[1] The Estimated Ancillary Plant Infrastructure Scale Factor represents the additional capital cost associated with constructing waste receiving equipment (e.g., loading range and material handling equipment), waste storage tank, ash treatment, and metal recovery processes (lime silos, material handling equipment, basins, etc.), utilities (boilers, electrical, water), process buildings, and administrative buildings.

Operating costs for the facility are assumed to include the following:

- The bulk of material processed is spent GAC, delivered at 50% water content by weight
- Labor for operation, assumed to require 30 full-time employees (FTE)
- Equipment maintenance, assumed to be 5% of the initial capital cost
- Natural gas use, based on the operation of similar facilities and throughput
- Electrical use, based on experience with similar facilities
- Ash disposal, assuming that 50% of the initial, dry GAC mass ends as ash
- Water costs, pretreatment costs, and WRRF surcharges for cooling tower blowdown, scrubber water, wash water from ash handling, and boiler water blowdown

Income for the facility would primarily be gained through incineration fees per mass of waste incinerated. While these costs vary by waste characteristics and will likely vary in the future, based on feedback from existing vendors and information summarized in U.S. EPA's interim guidance on PFAS disposal and destruction (U.S. EPA 2020), the estimated waste processing fee charged by existing facilities is approximately $1,300 per ton of waste received. Table 10-2 summarizes estimated annual O&M costs, income, and net profits for a regional high-temperature incineration facility.

Table 10-2 Annual estimated operational and maintenance costs and income for a regional high-temperature incineration facility handling spent GAC in Minnesota

Greenfield Incineration Facility	Unit	Unit Cost	Annual Quantity	Annual Cost[1]
Cost Analysis				
Staffing and administration	FTE	$75,000	30	($2,300,000)
Equipment maintenance	Percent of capital	$368,000,000	5%	($18,000,000)
Electrical use (for plant)	kWh	$0.09	19,000,000	($1,700,000)
Natural gas use (for combustion)	1,000 SCF	$8.14	2,900,000	($23,000,000)
Chemical use	tons	$170	1,000	($170,000)
Ash disposal	tons	$100	3,000[2]	($300,000)
Water costs, wastewater treatment, and surcharges[3]	$/gpm	$7,500	130	($980,000)
Other ancillary O&M costs not included here	Assumed 10% of items above			($4,800,000)
Total facility O&M Costs				**($52,000,000)**
Income analysis				
Income from incineration fees	tons wet spent media	$1,300	60,000	$78,000,000
Taxes on income	%	21%	$78,000,000	($16,000,000)
Total estimated annual facility net annual operating income				**$10,000,000**

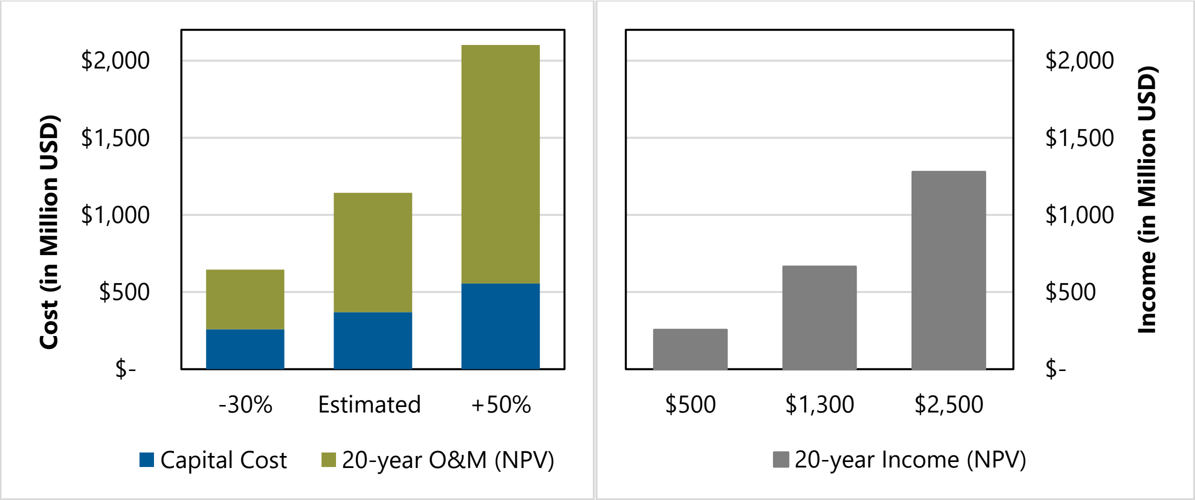
[1] Red text indicates expenses and black text indicates income. All annual costs subjected to +50%/-30% cost uncertainty range. Sums and products were rounded to two significant digits to reflect the uncertainty in the estimate, so some do not exactly match the sum or product of source values.

[2] Assumes 10% ash content of dry GAC and 50% initial water content of delivered material.

[3] Estimated water treatment and disposal costs of $1,500-$3,000/gpm for POTW discharge fees, plus $6,500-$10,000/gpm for pretreatment. Water treatment flow of approximately 130 gpm based on experience with similar facilities.

10.4.3 Economic Analysis

Twenty-year NPV costs are summarized in Figure 10-2. Generally, net annual income at $1,300/ton fees will be positive but may not be enough to pay back the initial capital investment. The payback time for the initial capital costs would likely be longer than 20 years, possibly closer to 50 years, which exceeds the typically-acceptable range for return on a large investment and may also exceed the operating lifetime of the facility. As a result, this project would likely require external financial support or higher fees, which may be possible given the lower anticipated transportation costs for a local facility summarized in Table 10-5.



Range of costs reflect the +50%/-30% range reflected in Tables 10-1 and 10-2.
Range of income reflects a range of potential disposal fees between $500 and $2,500 per ton.

Figure 10-2 Estimated 20-year NPV and income ranges for regional high-temperature incineration facility

Table 10-3 and Table 10-4 summarize the sensitivity analysis conducted for the regional facility concept. A range of 2%–6% real interest rate (achievable market rate interest, less inflation) was used to reflect a high degree of future uncertainty due to currently high inflation rates. A range was also used to reflect the potential fees the facility could charge for incineration, between $500/ton and $2,500/ton based on estimates for existing facility fees. The NPV of a 60,000 ton/year regional high-temperature incineration facility is the present value of the amortized capital cost and the sum of operation and maintenance costs over a specified evaluation period. An analysis of the sensitivity of the NPV to changes in tipping fee and real interest rate (Table 10-3) showed that the project is estimated to have a positive NPV if it can charge incineration fees over $2,000/ton. Using a transportation cost of $3/mile for 20-ton trucks ($100 savings per ton for decreasing trip length by 700 miles), this pricing is unlikely to be competitive, with out-of-state facilities currently charging approximately $1,300/ton. Table 10-4 illustrates NPV for different capital and O&M costs as well as fees and reflects the high level of uncertainty at this stage of analysis.

Table 10-3 Economic analysis for regional high-temperature incineration facility in Minnesota, in millions of USD—sensitivity of NPV to incineration fees and real interest rate

	Real interest rate	2%	2%	6%	6%	Simple payback period (years)
	Project evaluation period	10 years	20 years	10 years	20 years	
Incineration fees (USD per ton)	$500	-$620	-$830	-$580	-$690	No profit
	$1,000	-$420	-$450	-$410	-$430	No profit
	$1,500	-$200	-$59	-$230	-$150	19[1]
	$2,000	$16	$330	-$54	$120	9
	$2,500	$230	$730	$120	$400	6

[1] Some scenarios with negative 10-year or 20-year NPV have positive payback periods because the payback period does not include present value losses due to interest.

Table 10-4 Economic analysis for regional high-temperature incineration facility in Minnesota, in millions of USD—sensitivity to incineration fees and capital and O&M cost ranges (3% interest, 20 years)[1]

	Metric	NPV at 3% interest (Million USD)			Simple payback period (years)		
	Capital cost range	Low ($276)	Med ($368)	High ($460)	Low	Med	High
Incineration fees (USD per ton)	$500	-$300	-$790	-$1,700	No profit	No profit	No profit
	$1,000	$58	-$440	-$1,400	13	No profit	No profit
	$1,500	$410	-$86	-$1,000	6	19	No profit
	$2,000	$760	$270	-$690	4	9	No profit
	$2,500	$1,100	$620	-$330	3	6	32

[1] Cost range reflects both capital and operating cost ranges shown in Table 10-1 and Table 10-2.

Other items (externalities) considered in the economic analysis were benefits to Minnesota WRRFs, greenhouse gas impacts, and the potential for local air and noise pollution. The base case used for comparison assumes that the same amount of sorption media would be disposed of at existing, out-of-state regional high-temperature incineration facilities rather than at a new, in-state facility. Externality economics are summarized in Table 10-5. Local pollution externalities were not quantified but include economic costs associated with damage, decrease in land value, and loss of productivity; social costs such as medical costs; and environmental costs such as biodiversity and sustainability.

Table 10-5 Estimated economic externalities for regional high-temperature incineration in Minnesota (compared to out-of-state incineration)

Benefit externalities	Conversions used	Key metrics	Benefit or cost per year ($/year)
Trucking benefits for Minnesota WRRFs[1]	Trucking miles to cost	2,100,000 miles	$6,000,000
Greenhouse gas emissions benefits (decrease due to less trucking)[2]	Trucking ton-miles to cost Tons equivalent CO_2 to cost	42,000,000 ton-miles 7,500 tons equivalent CO_2	$400,000

[1] Benefits to WRRFs assume equal incineration fees for proposed versus existing out-of-state incinerators but a shorter shipping distance of 50 miles rather than 750 miles and $3/mile for 20-ton trucks. Total miles needed were estimated based on GAC use rates for Alternative 1a for municipal wastewater.

[2] GHG emission cost impacts assume the same energy use for incineration for the proposed versus existing out-of-state incinerator, with GHG savings due to shorter shipping distance based on the International Monetary Fund value of $50/ton (Gillingham 2019) with 161.8 grams of CO_2 per ton-mile of trucking, based on the Environmental Defense Fund (Mathers 2015).

The primary conclusion of the cost analysis of the regional high-temperature incineration facility is that tipping fees need to be over $1,500 to $2,000 (depending on final capital cost) to generate enough revenue over 20 years to offset the capital investment and operational costs.

10.5 Biosolids Pyrolysis/Gasification Facility Concept-Level Design

10.5.1 Design Basis and Equipment Needs

Outside of the Twin Cities metro area, most Minnesota WRRFs treat less than 20 MGD of wastewater influent and produce less than 20 dtpd of solids on average. A regional biosolids facility for PFAS destruction having a treatment capacity of 50 dtpd (200 wet tons of solids at 25% TS) could serve two or more local WRRFs in most areas of the state, except the Twin Cities metro region. Thus, 50 dtpd was selected as the biosolids treatment capacity for the regional pyrolysis/gasification conceptual design. Currently, two regional pyrolysis/gasification facilities in the United States can process more than 50 dtpd: one in New Jersey and the other in California.

A regional drying and pyrolysis/gasification facility requires the following:

- Biosolids receiving station and storage of biosolids
- Drying and pyrolysis/gasification treatment process (including conveyance, air treatment, and other ancillary processes as part of pyrolysis/gasification)
- Biochar storage
- Biochar loadout facility

An equipment vendor typically provides the treatment process as a complete package, including the dryer, pyrolyzer or gasifier, thermal oxidizer, and air emissions and handling equipment. Equipment and facilities supplied by the regional facility owner will be:

- Biosolids receiving station and conveyance
- Biosolids storage (short-term)
- Biochar storage and conveyance
- Biochar loadout
- Warehouse-style building to house dryer, pyrolyzer/gasifier, and thermal oxidizer
- Electrical, natural gas compressed air, and other supporting utilities to the drying and pyrolysis equipment

A summary of the key design parameters for the regional pyrolysis/gasification facility is provided in Table 10-6. For this study, the regional biosolids treatment facility was assumed to be constructed by one of the participating utilities on property already owned by the utility.

Table 10-6 Summary of design basis assumptions for regional biosolids pyrolysis/gasification facility

Design Parameter	Basis
Dewatered biosolids treatment capacity (dtpd)	50
Biosolids feedstock total solids (%TS)	25
Biosolids storage hopper volume (cubic feet, each)	6,400
Days of storage (total)	3
Receiving and hopper area footprint	100 ft x 50 ft
Biochar production rate (ton/dry ton of biosolids)	0.35-0.60
Biochar production (ton/day)	18-30
Days of storage of biochar	14
Biochar storage volume (cubic feet)	9,800
Dryer, pyrolysis/gasification, and thermal oxidizer system footprint	100 ft x 200 ft

A process flow diagram was developed to illustrate the concept of a regional drying and pyrolysis/gasification facility and is provided in Figure 10-3.

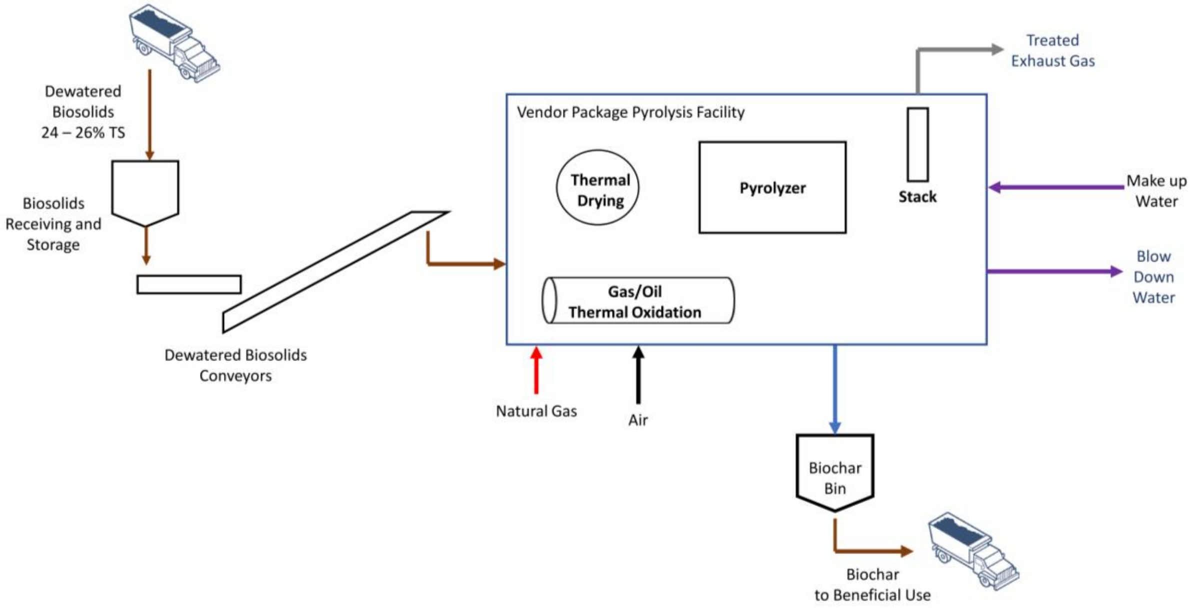


Figure 10-3 Conceptual process flow diagram for regional biosolids pyrolysis/gasification facility

10.5.2 Cost Estimates

Construction cost data for pyrolysis/gasification facilities handling from 4 dtpd to 50 dtpd were assembled and evaluated to develop the costs for a 50 dtpd regional facility (Figure 10-4). Cost estimates were prepared to Class 5 (AACE) accuracy. The estimates included all dryer and pyrolysis/gasification system equipment, piping and conveyance, product storage, ancillary systems, electrical, instrumentation, building, and indirect costs for constructing a complete PFAS destruction system for biosolids. Engineering fees were not included in the estimates.

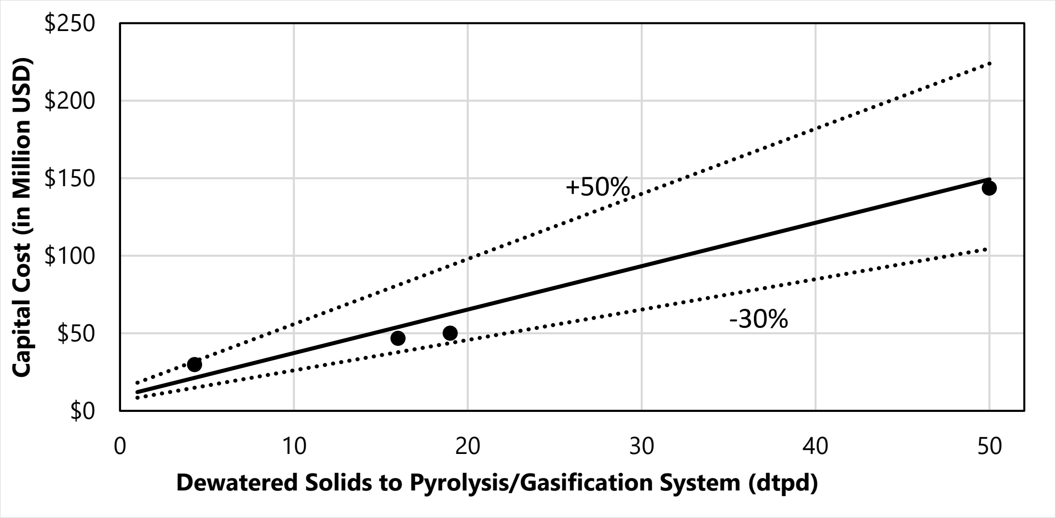


Figure 10-4 Construction cost in millions USD for a biosolids drying and pyrolysis/gasification facility with a capacity of up to 50 dtpd

A regional biosolids pyrolysis/gasification facility for PFAS destruction will likely require equipment in addition to what was included in the estimates used to develop the construction cost (Figure 10-4). Those items are:

- Biosolids receiving and conveyance
- Biosolids storage (short-term)
- Biochar storage and conveyance to loadout

For the conceptual 50 dtpd regional facility in this study, an additional $20 million was added to the capital cost estimate to account for the additional equipment. Table 10-7 summarizes the estimated capital costs for a regional biosolids pyrolysis/gasification facility.

Table 10-7 Estimated capital cost for a regional biosolids pyrolysis/gasification facility in Minnesota

Variable	Capital Cost
Biosolids pyrolysis/gasification process and facility, 50 dtpd	$140,000,000
Biosolids receiving, conveyance, and storage	$10,000,000
Biochar storage and loadout	$10,000,000
Total capital cost	$160,000,000 Range: (-30%) $110,000,000 to (+50%) $240,000,000

Operation and maintenance costs for the regional biosolids facility are assumed to include the following:

- Labor for operation and maintenance—assumed to require two FTEs
- Equipment maintenance—assumed to be 2% of initial capital cost
- Natural gas use at system startup
- Electricity input to the process

The operation and maintenance costs developed from three recent pyrolysis/gasification facility estimates and described in Section 7 of this Report can be applied to the regional facility. Estimates from pyrolysis/gasification systems larger than 10 dtpd showed a trend of decreasing costs per dry ton as the system size increased, as shown in Figure 10-5. A cost of $140 per dry ton was selected as a conservative O&M cost for a 50 dtpd regional facility based on existing installations, which translates to $2.56M in annual operating costs.

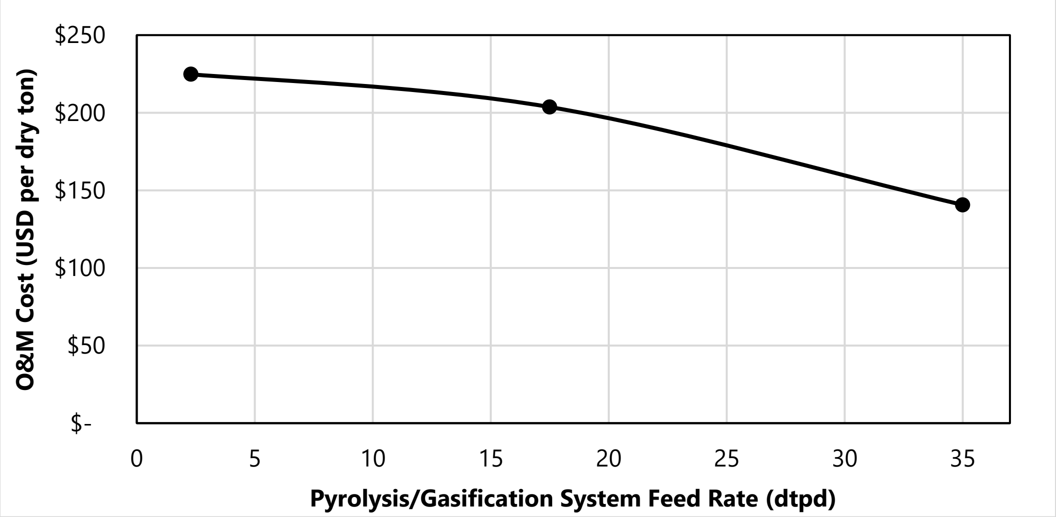


Figure 10-5 Estimated O&M cost per mass for biosolids pyrolysis/gasification with thermal oxidation

10.5.3 Economic Analysis

The economics of any regional biosolids pyrolysis/gasification facility will be unique. Many factors will influence the economic analysis, including any initial investment of the consortium in the system purchase, inflation of utility and commodity prices, fees charged per ton of biosolids received at the facility, and revenue generated from the sale of the biochar product. To simplify the economic analysis of a regional pyrolysis/gasification system, the utility constructing the pyrolysis/gasification facility was assumed to pay the facility's initial capital cost, which would then be amortized over the analysis period. Income would be obtained from the tipping fees on the biosolids received at the pyrolysis/gasification facility. The biochar product would not generate revenue or have a disposal cost. Energy recovered from the pyrolysis/gasification process will be fully used within the process.

The NPV of a 50 dtpd regional pyrolysis/gasification system is the present value of the amortized capital cost and the sum of operation and maintenance costs over a specified evaluation period. An analysis of the sensitivity of the NPV to changes in tipping fee and real interest rate is shown in Table 10-8. (This analysis demonstrates that the NPV is more sensitive to changes in real interest rate as the tipping fees increases. The scenarios with lower interest rates and the highest tipping fee charged over 20 years were the only scenarios with a positive NPV.

Table 10-8 Economic analysis for regional pyrolysis/gasification facility in Minnesota, in millions of USD—sensitivity of NPV to biosolids tipping fees and real interest rate

Real interest rate		2%	4%	6%	Simple payback period (years)
Project evaluation period		20 years	20 years	20 years	
Tipping fee (USD per wet ton)	$50	-$147	-$150	-$153	>20
	$100	-$86	-$100	-$110	>20
	$150	-$26	-$49	-$68	>20
	$200	$35	$1	-$25	15

A second analysis fixed the interest rate to test the sensitivity of the NPV and payback period to the capital cost and tipping fees, as shown in Table 10-9. The lowest capital cost and highest tipping fee scenarios achieved a payback period of fewer than 20 years. Half of the scenarios, however, required more than 20 years to achieve a simple payback on the system and its operation.

Table 10-9 Economic analysis for regional pyrolysis/gasification facility in Minnesota, in millions of USD - sensitivity to tipping fees and capital and O&M cost ranges (3% interest, 20 years)

Metric		NPV at 3% interest (Million USD)			Simple payback period (years)		
Capital cost range		Low ($110)	Med ($160)	High ($240)	Low	Med	High
Tipping fee ($/wet ton)	$50	-$97	-$150	-$233	>20	>20	>20
	$100	-$41	-$94	-$178	>20	>20	>20
	$150	$13	-$40	-$123	10	>20	>20
	$200	$38	$15	-$68	1	10	>20

While Table 10-9 indicates the 20-year NPV of the conceptual regional pyrolysis/gasification facility is favorable only at the higher tipping fee and lower construction cost range, it is important to consider that costs and revenue streams were kept simplistic for this initial evaluation of conceptual lifecycle costs. This analysis assumed the owner of the regional facility would bear the full cost of construction and maintenance over the 20-year period, collecting revenue only from tipping fees.

A regional facility could decrease its capital cost and increase annual income by contracting with the individual WRRFs to contribute to the construction cost of the facility and to pay an annual O&M fee. A scenario was developed for a 50 dtpd regional facility that included capital cost contributions from the participating utilities, annual fees, and tipping fees. It was assumed the regional facility operated at 80%

capacity on average. Biochar revenue was not included because its market value is currently unknown. The parameters used and the resulting NPV of the project for the regional facility are shown in Table 10-10. This scenario demonstrates the financial benefit to the regional facility owner of cost-sharing for the construction of the facility and charging regular maintenance fees.

Table 10-10 NPV of a regional pyrolysis/gasification facility in Minnesota with capital cost and O&M contributions from individual WRRFs

Variable	Value
Regional pyrolysis/gasification facility treatment capacity (dtpd)	50
Regional pyrolysis/gasification facility construction cost	-$160,000,000
Number of utilities using the regional pyrolysis/gasification facility	5
Construction cost contribution per utility	$5,000,000
Annual maintenance fee per utility	$1,000,000
Tipping fee collected (per wet ton)	$150/wet ton
Dewatered biosolids processed (wet ton per day)	160
Real interest rate	3%
NPV (3% interest, 20 years)	$28,800,000

This scenario demonstrates that a regional pyrolysis/gasification facility is economically viable for the owner of the regional facility when participating utilities pay more than the tipping fees alone. The scenario was further developed to compare the costs to a utility for contracting with the regional pyrolysis/gasification facility to the cost of constructing and operating their own pyrolysis/gasification facility. The utility was assumed to construct a 10 dtpd pyrolysis/gasification facility on their existing site and use it to process an average of 8 dtpd. In Table 10-11, costs detailed in Section 7 of this Report for constructing and operating an independent, 10 dtpd pyrolysis/gasification facility are compared to the cost to the utility for being part of a regional facility.

Table 10-11 Comparison of cost to utility for construction and operation of an independent pyrolysis/gasification facility versus a regional facility

	Value to utility - independent facility	Value to utility – regional facility
Pyrolysis/gasification treatment capacity on-site (dtpd)	10	0
Dewatered biosolids processed/trucked (dtpd)	8	8
Dewatered biosolids processed (wet ton per day, 25% TS)	32	32
Utility contribution to pyrolysis/gasification facility construction cost	-$75,700,000	-$5,000,000
Power, natural gas, labor, and consumables annual cost (in year 1, cost escalated over 20 years)	-$600,000	$0
Annual maintenance fee for regional facility	$0	-$1,000,000
Tipping fee (per wet ton)	$0	-$150/wet ton
Biosolids hauling cost ($3/mile, 50 miles round trip)[1]	$0	-$11/wet ton
NPV (3% interest, 20 years) for individual utility	-$87,600,000	-$41,600,000

[1] Full truck load is approximately 14 tons of biosolids. WRRF requires 2.3 trucks per day.

This analysis shows that pyrolysis/gasification treatment of biosolids would be a significant investment for a WRRF producing 10 dtpd of biosolids. However, the cost to the utility would be lower when participating in a consortium of smaller biosolids producers to support a regional pyrolysis/gasification facility.

The primary conclusions of the cost analysis of the regional pyrolysis/gasification facility are summarized into the following:

- Lower capital costs and a lower interest rates make the regional facility more economically viable.
- Tipping fees over $150/wet ton can generate enough revenue over 20 years to offset the capital investment and operational costs of a regional facility.
- The NPV of a regional pyrolysis/gasification facility is improved significantly if the participating utilities contribute to the initial construction cost and pay an annual maintenance fee.
- The cost to the utility for sending biosolids to the regional pyrolysis/gasification facility is less than half the cost of building and operating their own facility.
- A regional pyrolysis/gasification facility is an economically viable option if a group of utilities needs to implement biosolids treatment for PFAS destruction.

10.6 Permitting and Siting Considerations

Permitting and building a new PFAS-focused waste management facility in Minnesota present significant barriers. Currently, no PFAS destruction facilities are located in Minnesota. However, the permitting requirements for similar facilities provide a framework for what could be expected. For example, public acceptance of this type of facility could be a primary barrier and care would be needed during planning and siting to ensure that such a facility would not disproportionately affect disadvantaged communities.

The air emissions associated with a PFAS destruction facility are expected to be between 10 and 50 tons/year each for PM, SO_2, VOCs, and NO_x (Arkansas DEQ 2013). In addition, the potential for PFAS emissions from these facilities needs to be evaluated. Multiple studies have demonstrated over 99.9% removal of target PFAS, with airborne concentrations below detection limits (Thoma et al. 2022; DiStefano et al. 2022; Davis 2020), but a full mass balance on organic fluorine has not been closed. Minnesota has air emission standards for PFBA, PFHxA, PFHxS, PFOA, and PFOS in the range of 0.05 to 10 µg/m^3 (Minnesota Department of Health n.d.), all of which were met in the Arkansas study referenced above for a high-temperature incineration facility similar to the potential facility outlined here. However, airborne PFAS emissions were measured on the order of 0.35 µg /m^3 in stack gas for the sum of the five regulated compounds listed above (DiStefano et al. 2022), suggesting some potential to exceed Minnesota standards as designed.

This type of facility would be difficult to permit, and if permittable, would likely take many years to do so. A new facility would likely trigger a mandatory Environmental Impact Statement (EIS) pursuant to Minn. R. 4410.4400 Subpart 12 or 13 or a discretionary EIS if the project would be viewed as controversial by the public. An EIS would likely require a minimum of three years to approve and may take more than 10 years. The EIS would require several technical analyses, potentially including an unprecedented Air Emissions Risk Assessment focusing on PFAS emissions. The facility would likely trigger an air dispersion modeling assessment to demonstrate compliance with the National Ambient Air Quality Standards and may require additional air pollution control equipment (the primary concern being nitrous oxide emissions), which could further increase costs and permitting time. The facility may generate PFAS-containing industrial stormwater and wastewater streams and may require site-specific PFAS treatment processes required by the National Pollutant Discharge Elimination System (NPDES) program. Depending on the siting of the facility and public interest in the project, the facility location could present environmental justice concerns and/or undergo substantial public scrutiny during the public comment periods that would delay the permitting timeline.

Permitting would likely be easier for an existing service provider that already operates high-temperature incineration because existing suppliers would have actual emissions data from their operating units. For a new facility, several conservative assumptions (e.g., post-control toxics concentrations, pre-control storage tank specifications, etc.) could cause permitting issues and may require additional analyses (e.g., toxic pollutants sensitivity analyses, modeling iterations, sampling campaigns, etc.).

11 PFAS Management Themes and Conclusions

11.1 Breaking the Cycle

Future work to reduce the human and environmental health impacts of PFAS could focus on breaking the cycle of continuous shifting between soils, groundwater, surface water, and the atmosphere. Breaking the cycle could include reducing the load of PFAS routed to the environment in waste streams (the focus of this study), reducing cross-media transfer of PFAS, and removing PFAS from the environment for final destruction. Other upstream options could include producing fewer PFAS-containing products to limit future PFAS discharges to the environment.

As described in Section 1, wastewater and solid waste management systems are key routes for PFAS to enter the environment from a multitude of consumer products. These sources were illustrated in Figure 1-1. This study has evaluated the feasibility of currently available technologies and estimated the associated costs to apply these technologies for end-of-pipe PFAS removal and destruction for four waste streams, which would contribute to breaking the continued cycling of PFAS in the environment. Summary costs presented in this section generally use the lowest cost alternative for each waste stream. The alternative selected for liquid waste streams was tertiary treatment followed by GAC adsorption and GAC reactivation or incineration, depending on the facility flow rate. The alternative selected for WRRF biosolids was pyrolysis/gasification with thermal oxidation. However, each project should evaluate site-specific requirements and constraints to select the most suitable alternative.

11.2 Costs per Mass of PFAS Removed

The results of this study show that the cost per mass of PFAS destroyed is lower for higher-concentration waste streams like mixed MSW landfill leachate, compost contact water, and to some extent, WRRF biosolids, relative to municipal WRRF effluent.

Removal and destruction of PFAS over a 20-year operating period are expected to cost from $2.7 to $18 million per pound of PFAS removed for WRRF effluent (including retrofits for tertiary treatment) and $1.0 to $2.7 million per pound for WRRF biosolids (including biosolids dewatering upgrades) (Table 11-1). These costs will vary depending on facility size, biosolids production rates, and the concentration and speciation of PFAS reporting to biosolids and effluent at each facility. In contrast, the estimated cost per pound of PFAS removed is considerably lower at large landfill facilities, with potential costs as low as $400,000 per pound of PFAS removed at landfills treating 100 gpm of leachate.

This finding highlights the high cost of removing PFAS at WRRFs. Lower costs per mass of PFAS may be achieved at upstream facilities and landfills due to the higher concentration of PFAS in those discharges versus blended municipal WRRF influent. Costs per mass of PFAS removed at municipal WRRFs decrease as the facility size increases, indicating PFAS removal at larger WRRFs may be more economical. Costs per mass of PFAS removed are higher in compost contact water than in other waste streams evaluated due to the economy of scale for implementing complex water treatment at low flow rates.

Table 11-1 **Estimated cost per mass of PFAS removed from targeted waste streams over 20 years[1]**

	Size/Production		
Municipal WRRF facility size[3]	0.1 MGD	1 MGD	10 MGD
Municipal WRRF effluent capital	$7,300,000	$32,000,000	$120,300,000
Municipal WRRF effluent annual O&M	$500,000	$1,400,000	$6,400,000
Total 20-year cost for municipal WRRF effluent[2]	$12,600,000	$46,900,000	$188,200,000
Cost per lb PFAS removed over 20 years[5]	$18,100,000	$6,800,000	$2,700,000
Municipal WRRF biosolids production[4]	On-site biosolids management for facilities smaller than 1 dtpd is not expected to be economical.	1 dtpd	10 dtpd
Municipal WRRF biosolids capital		$24,600,000	$85,200,000
Municipal WRRF biosolids annual O&M		$200,000	$800,000
Total 20-year cost for municipal WRRF biosolids		$26,800,000	$93,700,000
Cost per lb PFAS removed over 20 years[5]		$2,700,000	$1,000,000
Mixed MSW landfill facility size[3]	1 GPM	10 GPM	100 GPM
Mixed MSW landfill leachate capital	$300,000	$800,000	$4,800,000
Mixed MSW landfill leachate annual O&M	$400,000	$400,000	$700,000
Total 20-year cost for mixed MSW landfills	$4,600,000	$5,100,000	$12,300,000
Cost per lb PFAS removed over 20 years[5]	$12,000,000	$1,400,000	$400,000
Composting facility size[3]	1 GPM	10 GPM	100 GPM
Compost contact water capital	$300,000	$800,000	$4,800,000
Compost contact water annual O&M	$300,000	$300,000	$600,000
Total 20-year cost for composting facilities	$3,500,000	$4,400,000	$11,200,000
Cost per lb PFAS removed over 20 years[5]	$39,300,000	$4,500,000	$1,300,000

MGD = million gallons per day, dtpd = dry tons per day, and gpm = gallons per minute

[1] Costs presented here reflect estimated project cost (Class 5 per AACE) developed with an associated uncertainty of +50%/-30% for both capital and annual O&M cost estimates. Costs are based on design basis concentrations selected to be typical of those reported in WRRF effluent (Helmer, Reeves, and Cassidy 2022; Coggan et al. 2019; Thompson et al. 2022), biosolids (Venkatesan and Halden 2013; Helmer, Reeves, and Cassidy 2022), landfill leachate (Lang et al. 2017), and compost contact water (Wood Environment & Infrastructure Solutions Inc. 2019). All costs are rounded up to the nearest $100,000.

[2] 20-year costs reflect present value calculations using an interest rate of 7%.

[3] Upgrade costs for liquid-phase treatment in WRRF effluent, mixed MSW landfill leachate, and compost contact water are for PFAS separation and destruction using GAC adsorption with high-temperature incineration of media (at flows below 10 MGD) or GAC reactivation (at 10 MGD or higher). These include approximate costs for tertiary treatment retrofits (at WRRFs) or pretreatment processes (at mixed MSW landfill leachate and composting sites) likely needed at most facilities to provide the water quality required for GAC feed.

[4] Upgrade costs are for PFAS destruction in WRRF biosolids using pyrolysis or gasification with thermal oxidation of produced gasses. Costs include centrifuge dewatering to provide 25% solids material for process feed for each facility.

[5] Mass PFAS removed reflect the sum of assumed concentrations of PFAS targeted in this study (PFOA, PFOS, PFHxA, PFHxS, PFBA, PFBS, 6:2 FTS, PFOSA, N-EtFOSAA, and N-MeFOSAA) as previously documented in Sections 2.3, 6.1.2, 7.1.2, 8.1.2, 9.1.2 multiplied by the flow rate or solids production rate shown.

11.3 PFAS Management Options for Minnesota

Society's ability to manage PFAS in the environment will depend upon strategies to remove these chemicals from consumer products and identify and implement cost-effective technologies to remove and destroy PFAS. Removal of PFAS from WRRF effluent using technologies currently available may not be practical due to the high costs per mass of PFAS treated. By comparison, the cost for removal of PFAS from WRRF biosolids, mixed MSW landfill leachate, and compost contact water is more efficient. Still, it could also benefit from innovation and improvement in the efficiency of future remediation technologies. Upstream source control measures may prove to be a more cost-effective approach to reducing PFAS than the PFAS management alternatives evaluated in this study.

The results of this study suggest that the 20-year cost to retrofit existing WRRFs larger than 0.05 MGD (typical for a town of about 500 people) to remove and destroy PFAS from WRRF effluent and biosolids across Minnesota is on the order of at least $14 billion for the PFAS listed in Table 2-1. Cost estimates for implementation at all plants larger than 0.05 MGD in Minnesota are summarized in Table 11-2. These cost estimates are order-of-magnitude estimates based on the estimated number and size of facilities in the state, using the treatment targets selected by MPCA and design bases defined in this Report. These costs will vary depending on the actual facilities implementing treatment.

The estimates for municipal WRRF biosolids given in Table 11-2 do not include the cost to upgrade the largest WRRF in Minnesota to pyrolysis/gasification with thermal oxidation. To date, there is no application of pyrolysis/gasification systems for biosolids at the scale of Minnesota's largest WRRF and no cost information available to estimate the 20-year cost for such a system. The largest facility might need to install multiple units on-site. The size and number of units could be determined as pyrolysis/gasification systems develop and scale up.

An exact count of mixed MSW landfills and SSOM compost facilities in Minnesota is difficult to develop due to mixed wastes accepted at many facilities. However, using an estimate of 24 mixed MSW landfills and nine SSOM compost facilities results in estimated 20-year costs of between $77M–$160M and $28M–$60M, respectively.

Table 11-2 Estimated 20-year costs for PFAS removal from targeted waste streams in Minnesota, in million USD[1]

Municipal WRRF facility size[3]	**0.1 MGD**	**1 MGD**	**10 MGD**	**300 MGD**	**Total for MN**
Approximate # of facilities in Minnesota	100	155	27	1	283
Municipal WRRF effluent capital costs	$500 - $1,100	$3,500 - $7,400	$2,300 - $4,900	$1,300 - $2,800	$7,500 - $16,000
Municipal WRRF effluent annual O&M costs	$29 - $60	$150 - $330	$120 - $260	$110 - $230	$410 - $870
Total 20-year costs, municipal WRRF effluent[2]	$800 - $1,700	$5,100 - $11,000	$3,600 - $7,600	$2,400 - $5,200	$12,000 - $25,000
Municipal WRRF biosolids production[4]	**<1 dtpd**	**1 dtpd**	**10 dtpd**		**Total for MN**
Approximate # of facilities in Minnesota	1 (regional)	40	10		51
Municipal WRRF biosolids capital costs	$110–$240	$680–$1,500	$630–$1,400		$1,400–$3,100
Municipal WRRF biosolids annual O&M costs	$2.5–$5.3	$4.7–$10	$5.5–$12		$13–$27
Total 20-year costs, municipal WRRF biosolids[2]	$140–$300	$730–$1,600	$690–$1,500		$1,600–$3,300
Mixed MSW landfill facility size[5]	**1 gpm**	**10 gpm**	**100 gpm**		**Total for MN**
Approximate # of facilities in Minnesota	6	15	3		24
Mixed MSW landfill leachate capital costs	$1.0–$3.0	$8.0–$18	$10–$20		$20–$40
Mixed MSW landfill leachate annual O&M costs	$1.3–$2.8	$3.4–$7.2	$0.74–$1.6		$5.4–$12
Total 20-year costs, mixed MSW landfills[2]	$15–$32	$44–$94	$18–$38		$77–$160
Composting facility size[6]	**1 gpm**	**10 gpm**	**100 gpm**		**Total for MN**
Approximate # of facilities in Minnesota	4	4	1		9
Compost contact water capital costs	$0.84–$1.8	$2.2–$4.8	$3.4–$7.2		$6.4–$14
Compost contact water annual O&M costs	$0.78–$1.7	$0.81–$1.7	$0.42–$0.90		$2.0–$4.3
Total 20-year costs, composting facilities[2]	$9.1–$20	$11–$23	$7.8–$17		$28–$60

MGD = million gallons per day, dtpd = dry tons per day, and gpm = gallons per minute

[1] This statewide evaluation carries additional uncertainty related to approximations made for facility sizing, number of facilities, and degree of pretreatment needed. Costs are rounded to two significant figures. Costs are based on design basis concentrations selected to be typical of those reported in WRRF effluent (Helmer, Reeves, and Cassidy 2022; Coggan et al. 2019; Thompson et al. 2022), biosolids (Venkatesan and Halden 2013; Helmer, Reeves, and Cassidy 2022), landfill leachate (Lang et al. 2017), and compost contact water (Wood Environment & Infrastructure Solutions Inc. 2019).

[2] Twenty-year costs reflect present value calculations using an interest rate of 7%.

[3] WRRF upgrade costs for effluent treatment are for PFAS separation and destruction using GAC adsorption with high-temperature incineration of media at flow rates below 1.1 MGD and GAC reactivation at higher flow rates. These include approximate costs for tertiary treatment retrofits (at WRRFs) or pretreatment processes (at mixed MSW landfill leachate and composting sites) likely needed at most facilities to provide the water quality required for GAC or RO feed. This analysis excludes WRRFs below 0.05 MGD.

[4] WRRF upgrade costs are for PFAS destruction in biosolids using pyrolysis or gasification with thermal oxidation of produced gasses. Costs include centrifuge dewatering to provide 25% solids material for process feed for each facility. These assume that WRRFs treating more than 0.1 MGD but producing less than 1 dtpd biosolids would ship to one regional, 50-dtpd pyrolysis facility. The costs shown here do not include transporting biosolids to that facility. These costs also do not include a pyrolysis/gasification facility with thermal oxidation for Minnesota's largest WRRF because costs for a facility of this size are not available.

[5] Operating landfill facilities in Minnesota and their size and leachate reported for Minnesota landfills accepting primarily MSW, assuming equalization is present to limit peak leachate flows to twice the annual average leachate flow.

[6] Operating compost facilities in Minnesota and their size were estimated based on publicly available data.

Costs developed here are meant to reflect order-of-magnitude costs based on facilities in Minnesota using GAC for liquid streams and pyrolysis for biosolids. The 300 MGD WRRF facility PFAS treatment costs were developed as a separate cost estimate using available equipment. For a GAC facility of this size, there are many design, operation, and maintenance uncertainties that were not explicitly considered in this conceptual cost.

In addition to the costs of building and operating a PFAS removal system, operator training and infrastructure would need to be enhanced to properly operate these systems, which have not been implemented for wastewater treatment in Minnesota. For context, wastewater operators have spent decades developing, refining, and passing along operational knowledge that allows conventional wastewater technologies to operate. A similar multi-year process of operational training and knowledge acquisition would take time to develop for PFAS-related technologies to work as designed.

While PFAS removal and destruction in the waste streams considered in this study may appear infeasible because of their estimated high costs, this study only considered technologies currently commercially available at the applicable scale. This presents an opportunity to support innovation and creative approaches to address PFAS treatment in these waste streams. These opportunities could include funding scale-up projects of emerging technologies, funding demonstration studies of emerging technologies for real-world waste streams, and funding comprehensive investigations of PFAS mass balances using both existing destruction technologies (i.e., high-temperature incinerators) and emerging technologies.

11.4 Energy Use and Associated Carbon Dioxide Emission Equivalents

While PFAS removal and destruction could be a future priority for waste and wastewater managers, sustainability and climate impacts are also increasingly prioritized. Table 11-3 summarizes estimated energy usage and CO_2 equivalents per mass of target PFAS removed from the liquid waste streams. Estimates are shown for the mid-range flows only for each waste stream. These estimates account for the following:

- The electricity and natural gas used specifically by PFAS separation equipment within the WRRF, landfill, or compost facility
- The electricity and natural gas used at the site or disposal facility (as applicable) to combust the sorption media concentrate

These estimates focus on energy usage from PFAS separation and destruction technologies. They do not account for CO_2 emissions associated with the production or transportation of the GAC media or AIX resin, mineralization of waste materials being destroyed (e.g., media), or beneficial reuse of treatment byproducts (e.g., use of regenerated GAC). The estimates also do not include energy usage associated with pretreatment ahead of the PFAS management alternatives. Other studies have evaluated the life cycle environmental impacts of PFAS treatment options in more detail (Boyer, Ellis, et al. 2021; G. Li et al. 2022).

The estimated weight of CO_2 equivalents emitted per weight of target PFAS removed ranges from 50 tons CO_2-eq/lb PFAS removed for foam fractionation of mixed MSW landfill leachate up to 2,200 lb CO_2-eq/lb PFAS removed for WRRF effluent treated with GAC and AIX resin and offsite high-temperature incineration of the sorption media. The CO_2 emission rates per pound of PFAS removed are highest for alternatives using GAC or a combination of GAC and AIX resin, at an estimated 1,000 and 2,200 ton CO_2-eq/lb PFAS for compost contact water and WRRF effluent, respectively. The exception is mixed MSW landfill leachate treated with GAC, where the CO_2 emission rate is 250 lb CO_2-eq/lb PFAS. The CO_2 emission rates per pound of PFAS removed are relatively low for mixed MSW landfill leachate due to the relatively high PFAS content of these waste streams. Foam fractionation with high-temperature incineration of the foamate for mixed MSW landfill leachate and compost contact water treatment both have lower estimated CO_2 emissions per pound of PFAS removed relative to GAC adsorption with high-temperature incineration of the media.

Table 11-3 Estimated energy use and CO_2 equivalents for PFAS management alternatives

Waste Stream	Facility Size	Highest Ranking Alternatives	Estimated Electricity Use (MWh/y) [1,2,3]	Estimated Natural Gas Use (kSCF/y) [1,2,3]	Estimated CO_2 Equivalent Emissions from Energy Use (ton/y)[4]	Estimated CO_2 Equivalents per Mass PFAS (ton CO_2/lb PFAS)[4,5]
Municipal WRRF effluent	1 MGD (700 gpm)	GAC with high-temperature incineration (Alt 1a)	210	10,000	720	2,100
		GAC and AIX with GAC and AIX high-temperature incineration (Alt 6a)	220	11,000	770	2,200
Mixed MSW landfill leachate	0.014 MGD (10 gpm)	GAC with high-temperature incineration (Alt 1a)	70	250	50	250
		Foam fractionation with high-temperature incineration of foamate (Alt 8a)	20	40	10	50
Compost contact water	0.014 MGD (10 gpm)	GAC with high-temperature incineration (Alt 1a)	70	210	40	1,000
		Foam fractionation with high-temperature incineration of foamate (Alt 8a)	20	40	10	230

[1] Electricity and natural gas use reflect estimates based on preliminary sizing and operational needs developed for this study and are subject to the +50%/-30% uncertainty range used throughout for operational costs.

[2] Estimates include the electricity and natural gas used specifically by the PFAS management systems within the WRRF, landfill, or compost facility, as well as the electricity and natural gas used at the disposal facility (as applicable) to combust the sorption media or concentrate. These estimates do not account for CO_2 emissions associated with media production, transportation, destruction of the media or biosolids materials, or potential emissions associated with the beneficial reuse of treatment byproducts (e.g., use of biochar from pyrolysis). The estimates also do not include energy usage associated with pretreatment ahead of the PFAS management alternatives.

[3] The amount of natural gas and electricity used by a high-temperature incinerator was estimated, assuming 46 MMBTU/ton of waste and 209 kWh/ton of waste, respectively, based on similar project experience.

[4] CO_2 emissions were estimated based on U.S. EPA guidance values of 953.7 lbs CO_2 per MWh and 55.1 kg CO_2 per 1,000 cubic feet of natural gas (U.S. EPA 2022a).

[5] Mass PFAS removed reflects the sum of assumed concentrations of PFAS targeted in this study (PFOA, PFOS, PFHxA, PFHxS, PFBA, PFBS, 6:2 FTS, and N-EtFOSAA) as previously documented in Sections 2.3, 6.1.2, 7.1.2, 8.1.2, 9.1.2 multiplied by the flow rate shown.

[6] Estimated energy use and CO_2 equivalents for WRRF biosolids treatment alternatives were excluded from this analysis. The energy consumption/production and CO_2 equivalents are unique to each biosolids installation and not suitable for this analysis.

11.5 Limitations on Use of Cost Estimates

Preliminary designs and costs were developed for a range of facility sizes to address design basis influent water and biosolids quality established for this study. Equipment sizing, media changeout frequency, and other factors are expected to vary across facilities depending on site-specific conditions and actual design bases, potentially pushing capital and O&M costs outside the range estimated for a specific design basis in this study. The costs presented here are intended to be useful for high-level, regional, and economic comparisons and are not appropriate for site-specific facility planning. Site-specific factors expected to influence actual technology selection, design, and costs include the following:

- Existing site infrastructure and treatment processes
- Space constraints and availability at the site
- Continuous versus intermittent nature of flows
- Water quality compared to design basis assumed
- Water quality variation
- PFAS concentrations and speciation
- Treatment goals for PFAS (concentrations, specific compounds, and reliability targets)
- Treatment goals for other parameters requiring removal
- Preferred site trade-offs for capital cost versus the level of operational complexity

11.6 Costs to Manage Short-Chain versus Long-Chain PFAS

Changing PFAS treatment targets and regulations complicate the estimation of treatment costs, especially O&M costs associated with sorption media changeout. While PFOA and PFOS have received the most regulatory attention, they are also two of the easier PFAS to remove from water because they are relatively hydrophobic. In contrast, short-chain PFAS, such as PFBA, PFHxA, and PFBS, prefer to stay in the water phase and are harder to remove through media sorption, AIX, and foam fractionation.

While 28 states have water phase guidelines (including drinking water, groundwater, surface water, and wastewater) for PFOA or PFOS, as of January 2023, only 14 states currently have water phase guidelines for PFBA or PFBS (ITRC 2023). The U.S. EPA’s recently proposed National Primary Drinking Water Regulation for PFAS includes PFOA, PFOS, PFNA, PFHxS, PFBS, and HFPO-DA (U.S. EPA 2023). This list also favors long-chain compounds over short-chain compounds and more sorbable sulfonates over alkyl acids. The availability of appropriate water treatment and environmental remediation technologies should be considered as the types of PFAS in use evolve, and regulations around use and environmental discharge are updated.

Alternatives relying on GAC sorption or AIX media to separate PFAS from the water phase will require much more frequent sorption media changeout to remove short-chain compounds consistently. This results in higher O&M costs for facilities using GAC or AIX media to remove short-chain PFAS, especially at higher flow rates. Figure 11-1 illustrates the estimated O&M cost multiplier for targeting short-chain PFAS (PFBA or PFBS) versus targeting long-chain PFAS (PFOA). This multiplier varies between a factor of 1.1 and 1.7 and increases with increasing flow. Both indicate that media changeout increasingly affects O&M costs at relatively high flows, high influent PFAS concentrations, and a high ratio of short-chain to long-chain PFAS concentrations.

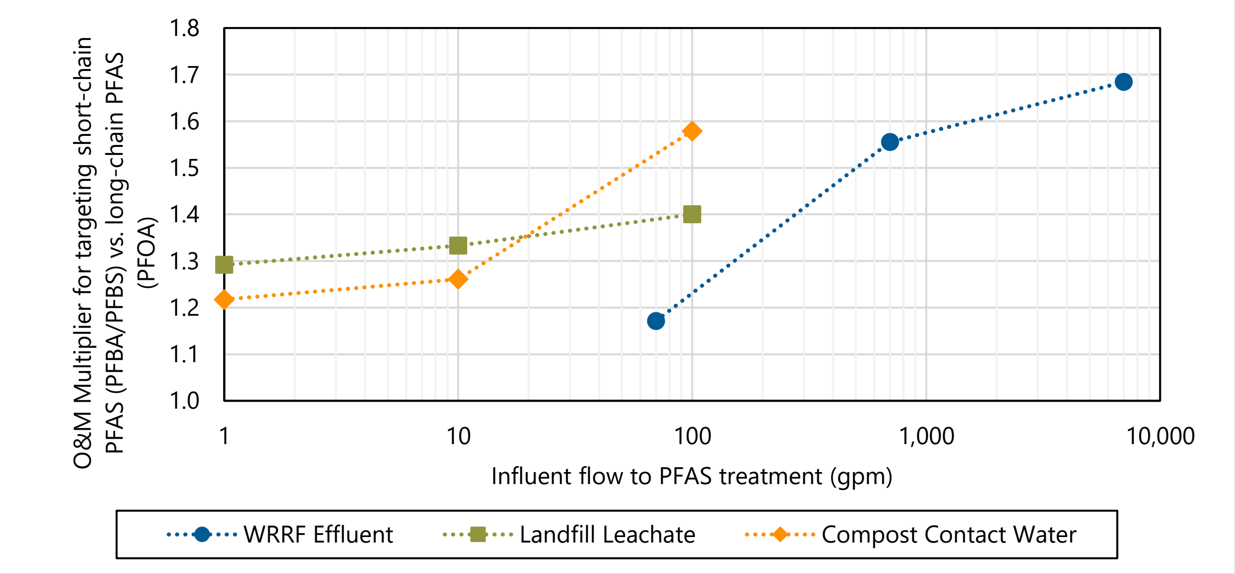


Figure 11-1 **Estimated O&M cost multiplier for GAC treatment targeting short-chain (PFBA or PFBS) removal versus long-chain (PFOA) removal for municipal WRRF effluent, mixed MSW landfill leachate, and compost contact water**

11.7 Technology Readiness

Except for foam fractionation, liquid treatment technologies currently available at commercial scales (i.e., GAC, AIX, modified clay, and RO) are considered general water treatment options; they remove many parameters and are not specifically designed for PFAS removal. More targeted technologies to further concentrate and destroy PFAS are being developed and have been demonstrated at bench and pilot scales. These technologies (including electrochemical oxidation, electrochemical or UV-assisted reduction, HALT, and SCWO) could potentially provide lower-cost and better implementable treatment at individual facilities, perhaps reducing or eliminating the need for regional or out-of-state high-temperature incineration facilities. However, due to investment and staffing challenges, the capacity of these technology vendors to scale-up equipment production is limited. Other challenges related to the evaluation of emerging PFAS destruction technologies include the following:

- Limited data on PFAS destruction efficacy at real sites with a full fluorine mass balance
- Potential for PFAS to escape with the gas phase
- Potential to produce alternate PFAS or other byproducts

- Ability to remove short-chain compounds
- Potential for scaling at high temperatures with salty feed streams
- High energy costs of some technologies, especially thermal technologies
- Regulatory uncertainty related to permitting installations and their emissions and discharges
- Ability to scale-up unit production, depending on future demand

Optimizing general destruction technologies for PFAS destruction continues to progress and needs additional scale-up opportunities to vet and optimize performance across a range of water quality and waste streams.

11.8 Future Research Needs

For PFAS management in the waste streams presented in this study to keep up with evolving regulations and public perception, the following research and development topics could be considered:

- Develop and apply regulatory pathways to permit facilities specifically for PFAS destruction
- Increase public and private investment in next-generation, lower-cost technologies to accelerate scale-up and implementation timelines
- Study full fluorine balance for next-generation PFAS separation and destruction technologies to understand destruction pathways and potential byproducts
- Conduct further research and support the development of low-temperature PFAS destruction options, with a specific focus on reducing full-scale treatment costs
- Consider safety concerns associated with PFAS aerosolization in existing and future processes

12 References

AACE International. 2020. "Cost Estimate Classification System." AACE International Recommended Practice No. 17R-97.

Appleman, Timothy D., Eric R.V. Dickenson, Christopher Bellona, and Christopher P. Higgins. 2013. "Nanofiltration and Granular Activated Carbon Treatment of Perfluoroalkyl Acids." Journal of Hazardous Materials 260 (September): 740–46. https://doi.org/10.1016/j.jhazmat.2013.06.033.

Appleman, Timothy D., Christopher P. Higgins, Oscar Quiñones, Brett J. Vanderford, Chad Kolstad, Janie C. Zeigler-Holady, and Eric R. V. Dickenson. 2014. "Treatment of Poly- and Perfluoroalkyl Substances in U.S. Full-Scale Water Treatment Systems." Water Research 51: 246–55. https://doi.org/10.1016/j.watres.2013.10.067.

Arkansas DEQ. 2013. "Statement of Basis for Issuance of Draft Air Permit #1009-AOP-R11, Clean Harbors El Dorado Facility." Arkansas Department of Environmental Quality.

AWWA. 2019. "Per- and Polyfluoroalkyl Substances (PFAS) - Treatment." www.awwa.org. https://www.awwa.org/Portals/0/AWWA/ETS/Resources/Per-andPolyfluoroalkylSubstances(PFAS)-Treatment.pdf?ver=2019-08-14-090249-580.

Bagg, W. K. 2009. "Infrastructure Optimisation via MBR Retrofit: A Design Guide." Water Science and Technology: A Journal of the International Association on Water Pollution Research 59 (2): 323–30. https://doi.org/10.2166/wst.2009.850.

Barranco, Frank, Paul Caprio, and George Hay. 2020. "Evaluation of Indirect Thermal Desorption Coupled with Thermal Oxidation Technology to Treat Solid PFAS-Impacted Investigation-Derived Waste." SERDP Project ER18-1572. Strategic Environmental Research and Development Program (SERDP). https://www.serdp-estcp.org/Program-Areas/Environmental-Restoration/ER18-1572.

Bates, Wayne T., Craig Bartels, and Rich Franks. 2008. "Improvements in Spiral Wound RO and NF Membrane and Element Construction for High Fouling Feed Water Applications." In . Hydranautics. https://membranes.com/knowledge-center/technical-papers/.

Beattie, Jessica, Mark Salvetti, and Tamzen Macbeth. 2022. "Surface Active Foam Fractionation Treatment Demonstration." CDM Smith. https://www.cdmsmith.com/en/Campaigns/SAFF-Pilot-Results.

Beecher, Ned, Juliana Beecher, Janine Burke-Wells, Maile Lono-Batura, Nora Goldstein, Greg Kester, and Bill Toffey. 2022. "National Biosolids Data Project: Data on Biosolids Management in the United States." 2022. https://www.biosolidsdata.org/.

Bellona, Christopher, Jörg E. Drewes, Pei Xu, and Gary Amy. 2004. "Factors Affecting the Rejection of Organic Solutes during NF/RO Treatment--a Literature Review." Water Research 38 (12): 2795–2809. https://doi.org/10.1016/j.watres.2004.03.034.

Berg, Chelsea, Brian Crone, Brian Gullett, Mark Higuchi, Max J. Krause, Paul M. Lemieux, Todd Martin, et al. 2022. "Developing Innovative Treatment Technologies for PFAS-Containing Wastes." Journal of the Air & Waste Management Association 72 (6): 540–55. https://doi.org/10.1080/10962247.2021.2000903.

Biodegradable Products Institute. 2020. "Fluorinated Chemicals." 2020. https://bpiworld.org/Fluorinated-Chemicals.

Bolea, Nicole. 2022. ECT2 – Regenerable IX Discussion. Phone.

Boyer, Treavor H., Anderson Ellis, Yida Fang, Charles E. Schaefer, Christopher P. Higgins, and Timothy J. Strathmann. 2021. "Life Cycle Environmental Impacts of Regeneration Options for Anion Exchange Resin Remediation of PFAS Impacted Water." Water Research 207: 117798. https://doi.org/10.1016/j.watres.2021.117798.

Boyer, Treavor H., Yida Fang, Anderson Ellis, Rebecca Dietz, Youn Jeong Choi, Charles E. Schaefer, Christopher P. Higgins, and Timothy J. Strathmann. 2021. "Anion Exchange Resin Removal of Per- and Polyfluoroalkyl Substances (PFAS) from Impacted Water: A Critical Review." Water Research 200 (July): 117244. https://doi.org/10.1016/j.watres.2021.117244.

Brepols, Ch., E. Dorgeloh, F.-B. Frechen, W. Fuchs, S. Haider, A. Joss, K. de Korte, et al. 2008. "Upgrading and Retrofitting of Municipal Wastewater Treatment Plants by Means of Membrane Bioreactor (MBR) Technology." Desalination 231 (1–3): 20–26. https://doi.org/10.1016/j.desal.2007.11.035.

Brepols, Ch., H. Schäfer, and N. Engelhardt. 2010. "Considerations on the Design and Financial Feasibility of Full-Scale Membrane Bioreactors for Municipal Applications." Water Science and Technology 61 (10): 2461–68. https://doi.org/10.2166/wst.2010.179.

Brown & Caldwell. 2019. "Conceptual Leachate Treatment Scoping Study for New England Waste Services of Vermont (NEWSVT) Landfill." https://legislature.vermont.gov/Documents/2020/WorkGroups/House%20Natural/Contaminants/W~Matt%20Chapman~Conceptual%20Leachate%20Treatment%20Scoping%20Study~1-16-2020.pdf.

Buckley, Thomas, Xiaoyong Xu, Victor Rudolph, Mahshid Firouzi, and Pradeep Shukla. 2022. "Review of Foam Fractionation as a Water Treatment Technology." Separation Science and Technology 57 (6): 929–58. https://doi.org/10.1080/01496395.2021.1946698.

Burkhardt, Jonathan B., Nick Burns, Dustin Mobley, Jonathan G. Pressman, Matthew L. Magnuson, and Thomas F. Speth. 2022. "Modeling PFAS Removal Using Granular Activated Carbon for Full-Scale System Design." Journal of Environmental Engineering 148 (3): 04021086. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001964.

Burkhardt, Jonathan, Carel Vandermeyden, Nick Burns, Dustin Mobley, Craig Patterson, Rajiv Khera, Jonathan Pressman, and Thomas Speth. 2019. "Modeling Pilot-Scale GAC PFAS Adsorption for the Simulation of Full-Scale Performance and Costs." Presented at the AWWA Water Quality and Technology Conference, Dallas, TX.

Burns, David J., Helena M. Hinrichsen, Paul Stevenson, and Peter J.C. Murphy. 2022. "Commercial-scale Remediation of Per- and Polyfluoroalkyl Substances from a Landfill Leachate Catchment Using Surface-Active Foam Fractionation (SAFF)." Remediation Journal 32: 139–50.

Burns, David J., Paul Stevenson, and Peter J. C. Murphy. 2021. "PFAS Removal from Groundwaters Using Surface-Active Foam Fractionation." Remediation Journal 31 (4): 19–33. https://doi.org/10.1002/rem.21694.

Choi, Youn Jeong, Rooney Kim Lazcano, Peyman Yousefi, Heather Trim, and Linda S. Lee. 2019. "Perfluoroalkyl Acid Characterization in U.S. Municipal Organic Solid Waste Composts." Environmental Science & Technology Letters 6 (6): 372–77. https://doi.org/10.1021/acs.estlett.9b00280.

Coggan, Timothy L., Damien Moodie, Adam Kolobaric, Drew Szabo, Jeff Shimeta, Nicholas D. Crosbie, Elliot Lee, Milena Fernandes, and Bradley O. Clarke. 2019. "An Investigation into Per- and

Polyfluoroalkyl Substances (PFAS) in Nineteen Australian Wastewater Treatment Plants (WWTPs)." Heliyon 5 (8): 1–11. https://doi.org/10.1016/j.heliyon.2019.e02316.

Cousins, Ian T., Gretta Goldenman, Dorte Herzke, Rainer Lohmann, Mark Miller, Carla A. Ng, Sharyle Patton, et al. 2019. "The Concept of Essential Use for Determining When Uses of PFASs Can Be Phased Out." Environmental Science: Processes & Impacts 21 (11): 1803–15. https://doi.org/10.1039/C9EM00163H.

Crittenden, John C., Neil J. Hutzler, David G. Geyer, Jacqueline L. Oravitz, and Gary Friedman. 1986. "Transport of Organic Compounds With Saturated Groundwater Flow: Model Development and Parameter Sensitivity." Water Resources Research 22 (3): 271–84. https://doi.org/10.1029/WR022i003p00271.

Crittenden, John C., R. Rhodes Trussell, David W. Hand, Kerry J. Howe, and George Tchobanoglous. 2012. Water Treatment: Principles and Design. Third. John Wiley & Sons, Inc.

Dai, Xiaodong, Zongli Xie, Brian Dorian, Stephen Gray, and Jianhua Zhang. 2019. "Comparative Study of PFAS Treatment by UV, UV/Ozone, and Fractionations with Air and Ozonated Air." Environmental Science: Water Research & Technology 5 (11): 1897–1907. https://doi.org/10.1039/C9EW00701F.

Darlington, Ramona, Edwin Barth, and John McKernan. 2018. "The Challenges of PFAS Remediation." The Military Engineer 110 (712): 58–60.

Davis, Brandon. 2020. "Application of Aries Process Technology to the Problem of PFAS Contamination." Aries Clean Energy. https://ariescleantech.com/white-paper-pfas-application-of-aries-process-technology-to-the-problem-of-pfas-contamination/.

DeCarolis, James, Samer Adham, William R. Pearce, Zakir Hirani, Stephen Lacy, and Roger Stephenson. 2007. "Cost Trends of MBR Systems for Municipal Wastewater Treatment." Proceedings of the Water Environment Federation 2007 (15): 3407–18. https://doi.org/10.2175/193864707787973734.

DiStefano, Rebecca, Tony Feliciano, Richard A. Mimna, Adam M. Redding, and John Matthis. 2022. "Thermal Destruction of PFAS during Full-scale Reactivation of PFAS-laden Granular Activated Carbon." Remediation Journal 32 (4): 231–38. https://doi.org/10.1002/rem.21735.

Donovan, Michael. 2022. "Landfill Leachate Treatment: PFAS Removal with Fluoro-Sorb Flex." Presented at the Cetco Webinar, June 23.

EGLE. 2020. "SUMMARY REPORT: Initiatives to Evaluate the Presence of PFAS in Municipal Wastewater and Associated Residuals (Sludge/Biosolids) in Michigan." Michigan Department of the Environment, Great Lakes, and Energy. https://www.michigan.gov/documents/egle/wrd-pfas-initiatives_691391_7.pdf.

Electric Power Research Institute. 2013. "Electricity Use and Management in the Municipal Water Supply and Wastewater Industries." https://www.epri.com/research/products/000000003002001433.

Evich, Marina G., Mary J. B. Davis, James P. McCord, Brad Acrey, Jill A. Awkerman, Detlef R. U. Knappe, Andrew B. Lindstrom, et al. 2022. "Per- and Polyfluoroalkyl Substances in the Environment." Science 375 (6580): eabg9065. https://doi.org/10.1126/science.abg9065.

Focus Environmental Inc. 2020. "Thermal Oxidizer Performance Test Report: Chemours Company Fayetteville Works," March, 65.

Franke, Vera, Miriam Dorothea Schäfers, Johan Joos Lindberg, and Lutz Ahrens. 2019. "Removal of Per- and Polyfluoroalkyl Substances (PFASs) from Tap Water Using Heterogeneously Catalyzed

Ozonation." Environ. Sci.: Water Res. Technol. 5 (11): 1887–96. https://doi.org/10.1039/C9EW00339H.

Franke, Vera, Malin Ullberg, Philip McCleaf, Maria Wålinder, Stephan J. Köhler, and Lutz Ahrens. 2021. "The Price of Really Clean Water: Combining Nanofiltration with Granular Activated Carbon and Anion Exchange Resins for the Removal of Per- And Polyfluoralkyl Substances (PFASs) in Drinking Water Production." ACS ES&T Water 1 (4): 782–95. https://doi.org/10.1021/acsestwater.0c00141.

Gillingham, Kenneth. 2019. "The True Cost of Reducing Greenhouse Gas Emissions." International Monetary Fund: Finance and Deveopment, December 2019.

Goldenberg, Marat. 2022. "Clay Mineral PFAS Treatment Inquiry," July 12, 2022.

Guo, Rui, Won-Jin Sim, Eung-Sun Lee, Ji-Hyun Lee, and Jeong-Eun Oh. 2010. "Evaluation of the Fate of Perfluoroalkyl Compounds in Wastewater Treatment Plants." Water Research 44 (11): 3476–86. https://doi.org/10.1016/j.watres.2010.03.028.

Hamid, Hanna, Loretta Y. Li, and John R. Grace. 2018. "Review of the Fate and Transformation of Per- and Polyfluoroalkyl Substances (PFASs) in Landfills." Environmental Pollution 235: 74–84. https://doi.org/10.1016/j.envpol.2017.12.030.

Helmer, Ross W., Donald M. Reeves, and Daniel P. Cassidy. 2022. "Per- and Polyfluorinated Alkyl Substances (PFAS) Cycling within Michigan: Contaminated Sites, Landfills and Wastewater Treatment Plants." Water Research 210. https://doi.org/10.1016/j.watres.2021.117983.

Herrmann, Ronald, Carolyn Acheson, Larry Zintek, and Danielle Kleinmaier. 2020. "Land Application Field Study II." Presented at the US EPA Biosolids Webinar, Virtual, October 29. https://cfpub.epa.gov/si/si_public_file_download.cfm?p_download_id=538566&Lab=NRMRL.

Houtz, Erika F., Rebecca Sutton, June-Soo Park, and Margaret Sedlak. 2016. "Poly- and Perfluoroalkyl Substances in Wastewater: Significance of Unknown Precursors, Manufacturing Shifts, and Likely AFFF Impacts." Water Research 95: 142–49. https://doi.org/10.1016/j.watres.2016.02.055.

ITRC. 2018. "Remediation Technologies and Methods for Per- and Polyfluoralkyl Substances (PFAS)." ITRC. https://pfas-1.itrcweb.org/fact_sheets_page/pfas_fact_sheet_remediation_3_15_18.pdf.

———. 2020. "Naming Conventions and Physical and Chemical Properties of Per- and Polyfluoroalkyl Substances (PFAS)." Interstate Technology Regulatory Council. https://pfas-1.itrcweb.org/fact_sheets_page/pfas_fact_sheet_naming_conventions__4_8_20.pdf.

———. 2021. "Chemistry, Terminology, and Acronyms." Interstate Technology Regulatory Council: PFAS - Per- and Polyfluoroalkyl Substances. 2021. https://pfas-1.itrcweb.org/2-2-chemistry-terminology-and-acronyms/.

———. 2022. "PFAS Treatment Technologies." PFAS Technical and Regulatory Guidance Document and Fact Sheets. 2022. https://pfas-1.itrcweb.org/12-treatment-technologies/.

———. 2023. "PFAS Regulations, Guidance and Advisory Values." https://pfas-1.itrcweb.org/wp-content/uploads/2023/02/ITRCPFASWaterandSoilValuesTables_JAN2023-Final.xlsx.

Jarvie, Michelle Edith, David W. Hand, Shanmugalingam Bhuvendralingam, John C. Crittenden, and Dave R. Hokanson. 2005. "Simulating the Performance of Fixed-Bed Granular Activated Carbon Adsorbers: Removal of Synthetic Organic Chemicals in the Presence of Background Organic Matter." Water Research 39 (11): 2407–21. https://doi.org/10.1016/j.watres.2005.04.023.

Jiang, Tao, Weilan Zhang, Aswin Kumar Ilango, Jeremy I. Feldblyum, Zheng Wei, Haralabos Efstathiadis, Mehmet V. Yigit, and Yanna Liang. 2022. "Surfactant-Modified Clay for Adsorption of Mixtures of Per- and Polyfluoroalkyl Substances (PFAS) in Aqueous Solutions." ACS Applied Engineering Materials, October, acsaenm.2c00096. https://doi.org/10.1021/acsaenm.2c00096.

Jin, Tonghui, Mohammad Peydayesh, and Raffaele Mezzenga. 2021. "Membrane-Based Technologies for per- and Poly-Fluoroalkyl Substances (PFASs) Removal from Water: Removal Mechanisms, Applications, Challenges and Perspectives." Environment International 157 (December): 106876. https://doi.org/10.1016/j.envint.2021.106876.

Kim, Dong-Jin, Dong-Ig Lee, Gi-Cheol Cha, and Jurg Keller. 2008. "Analysis of Free Ammonia Inhibition of Nitrite Oxidizing Bacteria Using a Dissolved Oxygen Respirometer." Environmental Engineering Research 13 (3): 125–30. https://doi.org/10.4491/eer.2008.13.3.125.

Kjeldsen, Peter, Morton A. Barlaz, Alix P. Rooker, Anders Baun, Anna Ledin, and Thomas H. Christensen. 2002. "Present and Long-Term Composition of MSW Landfill Leachate: A Review." Critical Reviews in Environmental Science and Technology 32 (4): 297–336. https://doi.org/10.1080/10643380290813462.

Krause, Max J., Eben Thoma, Endalkachew Sahle-Damesessie, Brian Crone, Andrew Whitehill, Erin Shields, and Brian Gullett. 2022. "Supercritical Water Oxidation as an Innovative Technology for PFAS Destruction." Journal of Environmental Engineering 148 (2): 05021006. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001957.

Krogmann, Uta, and Heike Woyczechowski. 2000. "Selected Characteristics of Leachate, Condensate and Runoff Released during Composting of Biogenic Waste." Waste Management & Research 18 (3): 235–48. https://doi.org/10.1177/0734242X0001800305.

Kyser, Scott. 2022. Membrane Bioreactor Capital Costs Data for Minnesota Installations.

Lang, Johnsie R., B. McKay Allred, Jennifer A. Field, James W. Levis, and Morton A. Barlaz. 2017. "National Estimate of Per- and Polyfluoroalkyl Substance (PFAS) Release to U.S. Municipal Landfill Leachate." Environmental Science & Technology 51 (4): 2197–2205. https://doi.org/10.1021/acs.est.6b05005.

Lewis, FM. 2008. "Combustion Fundamentals." In Proceedings of the Water Environmental Federation Residuals and Biosolids Conference.

Li, Gengyang, Joshua Dunlap, Yifei Wang, Qingguo Huang, and Ke Li. 2022. "Environmental Life Cycle Assessment (LCA) of Treating PFASs with Ion Exchange and Electrochemical Oxidation Technology." ACS ES&T Water 2 (9): 1555–64. https://doi.org/10.1021/acsestwater.2c00196.

Li, Jianna, Conrad Austin, Stuart Moore, Brian R. Pinkard, and Igor V. Novosselov. 2023. "PFOS Destruction in a Continuous Supercritical Water Oxidation Reactor." Chemical Engineering Journal 451 (January): 139063. https://doi.org/10.1016/j.cej.2022.139063.

Lin, Diana, Miguel Mendez, and Rebecca Sutton. 2021. "PFAS Study Phase 1 Results." Presented at the BACWA Board Meeting, Virtual, August 20. https://bacwa.org/document/sfei-pfas-phase-1-presentation-2021-09-01/.

Liu, Yen-Ling, and Mei Sun. 2021. "Ion Exchange Removal and Resin Regeneration to Treat Per- and Polyfluoroalkyl Ether Acids and Other Emerging PFAS in Drinking Water." Water Research 207 (December): 117781. https://doi.org/10.1016/j.watres.2021.117781.

Lo, C. H., E. McAdam, and S. Judd. 2015. "The Cost of a Small Membrane Bioreactor." Water Science and Technology 72 (10): 1739–46. https://doi.org/10.2166/wst.2015.394.

Lozowski, Dorothy. n.d. "The Chemical Engineering Plant Cost Index." Chemical Engineering (blog). Accessed November 9, 2022. https://www.chemengonline.com/pci-home/.

Mathers, Jason. 2015. "Green Freight Math: How to Calculate Emissions for a Truck Move." Environmental Defense Fund+Business (blog). March 24, 2015. https://business.edf.org/insights/green-freight-math-how-to-calculate-emissions-for-a-truck-move/.

McNamara, James D., Ralph Franco, Richard Mimna, and Leo Zappa. 2018. "Comparison of Activated Carbons for Removal of Perfluorinated Compounds From Drinking Water: Comparison of Activated Carbons for Removal of Perfluorinated Compounds From Drinking Water." Journal - American Water Works Association 110 (1): E2–14. https://doi.org/10.5942/jawwa.2018.110.0003.

Meng, Pingping, Shubo Deng, Ayiguli Maimaiti, Bin Wang, Jun Huang, Yujue Wang, Ian T. Cousins, and Gang Yu. 2018. "Efficient Removal of Perfluorooctane Sulfonate from Aqueous Film-Forming Foam Solution by Aeration-Foam Collection." Chemosphere 203: 263–70. https://doi.org/10.1016/j.chemosphere.2018.03.183.

Minnesota Department of Health. n.d. "Air Guidance Values." Accessed October 7, 2022. https://www.health.state.mn.us/communities/environment/risk/guidance/air/table.html.

Minnesota Pollution Control Agency. 2022. "Composting and Managing Organic Waste." 2022. https://www.pca.state.mn.us/air-water-land-climate/composting-and-managing-organic-waste.

MPCA. 2020. "Guidance for Perfluorochemicals Analysis." Minnesota Pollution Control Agency. https://www.pca.state.mn.us/sites/default/files/p-eao2-06.pdf.

NGWA. 2017. "Groundwater and PFAS: State of Knowledge and Practice." Westville: NGWA Press.

Nian, Min, Kai Luo, Fei Luo, Ruxianguli Aimuzi, Xiaona Huo, Qian Chen, Ying Tian, and Jun Zhang. 2020. "Association between Prenatal Exposure to PFAS and Fetal Sex Hormones: Are the Short-Chain PFAS Safer?" Environmental Science & Technology 54 (13): 8291–99. https://doi.org/10.1021/acs.est.0c02444.

Niessen, Walter R. 2002. Combustion and Incineration Processes. 3rd ed., rev.Expanded. Environmental Science and Pollution Control Series 25. New York: Marcel Dekker.

Pan, Chang-Gui, You-Sheng Liu, and Guang-Guo Ying. 2016. "Perfluoroalkyl Substances (PFASs) in Wastewater Treatment Plants and Drinking Water Treatment Plants: Removal Efficiency and Exposure Risk." Water Research 106: 562–70. https://doi.org/10.1016/j.watres.2016.10.045.

Pannu, Meeta, and Megan Plumlee. 2021. "PFAS Phase I Pilot-Scale Treatment Study Final Report." Orange Couty Water District. https://www.ocwd.com/media/9829/2021-03-24_ocwd-pfas-pilot-i_finalreport.pdf.

Paz-Ferreiro, Jorge, Aurora Nieto, Ana Méndez, Matthew Peter James Askeland, and Gabriel Gascó. 2018. "Biochar from Biosolids Pyrolysis: A Review." International Journal of Environmental Research and Public Health 15 (5): 956. https://doi.org/10.3390/ijerph15050956.

Pinkard, Brian R., Shreyas Shetty, Derek Stritzinger, Christopher Bellona, and Igor V. Novosselov. 2021. "Destruction of Perfluorooctanesulfonate (PFOS) in a Batch Supercritical Water Oxidation Reactor." Chemosphere 279: 130834. https://doi.org/10.1016/j.chemosphere.2021.130834.

Robey, Nicole M., Bianca F. da Silva, Michael D. Annable, Timothy G. Townsend, and John A. Bowden. 2020. "Concentrating Per- and Polyfluoroalkyl Substances (PFAS) in Municipal Solid Waste Landfill Leachate Using Foam Separation." Environmental Science & Technology 54 (19): 12550–59. https://doi.org/10.1021/acs.est.0c01266.

Roy, Dany, Antonin Azaïs, Sanae Benkaraache, Patrick Drogui, and Rajeshwar D. Tyagi. 2018. "Composting Leachate: Characterization, Treatment, and Future Perspectives." Reviews in Environmental Science and Bio/Technology 17 (2): 323–49. https://doi.org/10.1007/s11157-018-9462-5.

Schaefer, Charles E., Dina M. Drennan, Danielle N. Tran, Raymmah Garcia, Emerson Christie, Christopher P. Higgins, and Jennifer A. Field. 2019. "Measurement of Aqueous Diffusivities for Perfluoroalkyl Acids." Journal of Environmental Engineering 145 (11): 06019006. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001585.

Siriwardena, Dinusha P., Ryan James, Kavitha Dasu, Jonathan Thorn, Ramona Darlington Iery, Franco Pala, Denise Schumitz, Stephanie Eastwood, and Nathan Burkitt. 2021. "Regeneration of Per- and Polyfluoroalkyl Substance-Laden Granular Activated Carbon Using a Solvent Based Technology." Journal of Environmental Management 289: 112439. https://doi.org/10.1016/j.jenvman.2021.112439.

Smith, Sanne J., Karin Wiberg, Philip McCleaf, and Lutz Ahrens. 2022. "Pilot-Scale Continuous Foam Fractionation for the Removal of Per- and Polyfluoroalkyl Substances (PFAS) from Landfill Leachate." ACS ES&T Water 2 (5): 841–51. https://doi.org/10.1021/acsestwater.2c00032.

Sonmez Baghirzade, Busra, Yi Zhang, James F. Reuther, Navid B. Saleh, Arjun K. Venkatesan, and Onur G. Apul. 2021. "Thermal Regeneration of Spent Granular Activated Carbon Presents an Opportunity to Break the Forever PFAS Cycle." Environmental Science & Technology 55 (9): 5608–19. https://doi.org/10.1021/acs.est.0c08224.

Stanford, Patrick. 2019. "Leachate Treatment Technologies for PFAS." Presented at the Federation of New York Solid Waste Association. https://nyfederation.org/wp-content/uploads/2019/pdf2019/54%20Stanford%20P.pdf.

Steinle-Darling, Eva, and Martin Reinhard. 2008. "Nanofiltration for Trace Organic Contaminant Removal: Structure, Solution, and Membrane Fouling Effects on the Rejection of Perfluorochemicals." Environmental Science & Technology 42 (14): 5292–97. https://doi.org/10.1021/es703207s.

Stoiber, Tasha, Sydney Evans, and Olga V. Naidenko. 2020. "Disposal of Products and Materials Containing Per- and Polyfluoroalkyl Substances (PFAS): A Cyclical Problem." Chemosphere 260 (December): 127659. https://doi.org/10.1016/j.chemosphere.2020.127659.

Tang, Chuyang Y., Q. Shiang Fu, Craig S. Criddle, and James O. Leckie. 2007. "Effect of Flux (Transmembrane Pressure) and Membrane Properties on Fouling and Rejection of Reverse Osmosis and Nanofiltration Membranes Treating Perfluorooctane Sulfonate Containing Wastewater." Environmental Science & Technology 41 (6): 2008–14. https://doi.org/10.1021/es062052f.

Thoma, Eben D., Robert S. Wright, Ingrid George, Max Krause, Dario Presezzi, Valentino Villa, William Preston, Parik Deshmukh, Phil Kauppi, and Peter G. Zemek. 2022. "Pyrolysis Processing of PFAS-Impacted Biosolids, a Pilot Study." Journal of the Air & Waste Management Association (1995) 72 (4): 309–18. https://doi.org/10.1080/10962247.2021.2009935.

Thompson, Kyle A., Soroosh Mortazavian, Dana J. Gonzalez, Charles Bott, Jennifer Hooper, Charles E. Schaefer, and Eric R. V. Dickenson. 2022. "Poly- and Perfluoroalkyl Substances in Municipal Wastewater Treatment Plants in the United States: Seasonal Patterns and Meta-Analysis of Long-Term Trends and Average Concentrations." ACS ES&T Water 2 (5): acsestwater.1c00377. https://doi.org/10.1021/acsestwater.1c00377.

Tribe, Michael, and R.L.W. Alpine. 1986. "Scale Economies and the '0.6 Rule.'" Engineering Costs and Production Economics 10 (February): 271–78. https://doi.org/10.1016/S0167-188X(86)80025-8.

U.S. EIA. 2022a. "Electricity Data Browser." 2022. https://www.eia.gov/electricity/data/browser/.

———. 2022b. "Natural Gas." Natural Gas Cost Summary. 2022. https://www.eia.gov/dnav/ng.

U.S. EPA. 2020. "Interim Guidance on the Destruction and Disposal of Perfluoroalkyl and Polyfluoroalkyl Substances and Materials Containing Perfluoroalkyl and Polyfluoroalkyl Substances." https://www.epa.gov/system/files/documents/2021-11/epa-hq-olem-2020-0527-0002_content.pdf.

———. 2021a. "CompTox Chemicals Dashboard: Navigation Panel to PFAS Structure Lists." August 2021. https://comptox.epa.gov/dashboard/chemical-lists/PFASSTRUCT.

———. 2021b. "Draft Method 1633, Analysis of Per- and Polyfluoroalkyl Substances (PFAS) in Aqueous, Solid, Biosolids, and Tissue Samples by LC-MS/MS, EPA 821-R-21-001." United States Environmental Protection Agency PFAS Innovative Treatment Team. https://www.epa.gov/system/files/documents/2021-09/method_1633_draft_aug-2021.pdf.

———. 2021c. "Emerging Issues in Food Waste Management Persistent Chemical Contaminants." EPA/600/R-21/115. United States Environmental Protection Agency Office of Research and Development.

———. 2022a. "Greenhouse Gases Equivalencies Calculator - Calculations and References." Data and Tools. June 23, 2022. https://www.epa.gov/energy/greenhouse-gases-equivalencies-calculator-calculations-and-references.

———. 2022b. "National Overview: Facts and Figures on Materials, Wastes and Recycling." United States Environmental Protection Agency. https://www.epa.gov/facts-and-figures-about-materials-waste-and-recycling/national-overview-facts-and-figures-materials.

———. 2023. "Per- and Polyfluoroalkyl Substances (PFAS): Proposed PFAS National Primary Drinking Water Regulation." March 14, 2023. https://www.epa.gov/sdwa/and-polyfluoroalkyl-substances-pfas.

Venkatesan, Arjun K., and Rolf U. Halden. 2013. "National Inventory of Perfluoroalkyl Substances in Archived U.S. Biosolids from the 2001 EPA National Sewage Sludge Survey." Journal of Hazardous Materials 0: 413–18. https://doi.org/10.1016/j.jhazmat.2013.03.016.

Vo, Phong H. N., Thomas Buckley, Xiaoyong Xu, Thi Minh Hong Nguyen, Victor Rudolph, and Pradeep Shukla. 2023. "Foam Fractionation of Per- and Polyfluoroalkyl Substances (PFASs) in Landfill Leachate Using Different Cosurfactants." Chemosphere 310: 136869. https://doi.org/10.1016/j.chemosphere.2022.136869.

Wang, Jiaxuan, Lei Wang, Changqing Xu, Rui Zhi, Rui Miao, Tong Liang, Xianglei Yue, Yongtao Lv, and Tingting Liu. 2018. "Perfluorooctane Sulfonate and Perfluorobutane Sulfonate Removal from Water by Nanofiltration Membrane: The Roles of Solute Concentration, Ionic Strength, and Macromolecular Organic Foulants." Chemical Engineering Journal 332 (January): 787–97. https://doi.org/10.1016/j.cej.2017.09.061.

Wei, Zongsu, Tianyuan Xu, and Dongye Zhao. 2019. "Treatment of Per- and Polyfluoroalkyl Substances in Landfill Leachate: Status, Chemistry and Prospects." Environmental Science: Water Research & Technology 5 (11): 1814–35. https://doi.org/10.1039/C9EW00645A.

Westreich, Philippe, Richard Mimna, Jenalle Brewer, and Frederic Forrester. 2018. "The Removal of Short-Chain and Long-Chain Perfluoroalkyl Acids and Sulfonates via Granular Activated Carbons: A Comparative Column Study." Remediation Journal 29 (1): 19–26. https://doi.org/10.1002/rem.21579.

Winchell, Lloyd J., John J. Ross, Martha J. M. Wells, Xavier Fonoll, John W. Norton, and Katherine Y. Bell. 2021a. "Per- and Polyfluoroalkyl Substances Thermal Destruction at Water Resource Recovery Facilities: A State of the Science Review." Water Environment Research 93 (6): 826–43. https://doi.org/10.1002/wer.1483.

———. 2021b. "Per- and Polyfluoroalkyl Substances Thermal Destruction at Water Resource Recovery Facilities: A State of the Science Review." Water Environment Research 93 (6): 826–43. https://doi.org/10.1002/wer.1483.

Winchell, Lloyd, John Ross, John Norton, Xavier Fonoll, Majid Khan, and Stephen Kuplicki. 2022. "Fate of PFAS through Incineration at Wastewater Reclamation Facilities." In . Columbus, OH.

Wood Environment & Infrastructure Solutions Inc. 2019. "Investigation of Per- and Polyfluoroalkyl Substances (PFAS) at Select Source Separated Organic Material and Yard Waste Sites, Minnesota." Minnesota Pollution Control Agency.

Xiao, Feng, Pavankumar Challa Sasi, Bin Yao, Alena Kubátová, Svetlana A. Golovko, Mikhail Y. Golovko, and Dana Soli. 2020. "Thermal Stability and Decomposition of Perfluoroalkyl Substances on Spent Granular Activated Carbon." Environmental Science & Technology Letters, April. https://doi.org/10.1021/acs.estlett.0c00114.

Yan, Bei, Gabriel Munoz, Sébastien Sauvé, and Jinxia Liu. 2020. "Molecular Mechanisms of Per- and Polyfluoroalkyl Substances on a Modified Clay: A Combined Experimental and Molecular Simulation Study." Water Research 184 (October): 116166. https://doi.org/10.1016/j.watres.2020.116166.

Young, Thor, Sebastian Smoot, Jeff Peeters, and Pierre Côté. 2014. "Cost-Effectiveness of Membrane Bioreactors Treatment System for Low-Level Phosphorus Reduction from Municipal Wastewater." Water Practice and Technology 9 (3): 316–23. https://doi.org/10.2166/wpt.2014.033.

Zhao, Changwei, Jian Zhang, Guangzhi He, Tao Wang, Deyin Hou, and Zhaokun Luan. 2013. "Perfluorooctane Sulfonate Removal by Nanofiltration Membrane the Role of Calcium Ions." Chemical Engineering Journal 233 (November): 224–32. https://doi.org/10.1016/j.cej.2013.08.027.

Appendices

Appendix A

PFAS Chemical Characteristics and Most Similar "Target PFAS" Used in this Study

Appendix A PFAS Chemical Characteristics and Most Similar "Target PFAS" Used in this Study

Abbreviation	Name	CAS Number	Total No. C atoms	Number of fluorinated C atoms	MW	Fuctional group	Most Similar Target PFAS
PFMPA	Perfluoro-3-methoxypropanoic acid	377-73-1	4	3	230	COOH	PFBA
PFBA	Perfluorobutanoic acid	375-22-4	4	3	214	COOH	
PFMBA	Perfluoro-4-methoxybutanoic acid	151772-58-6	5	4	280	COOH	PFBA
PFPeA	Perfluoropentanoic acid	2706-90-3	5	4	264	COOH	PFHxA
NFDHA	Perfluoro-3,6-dioxaheptanoic acid	151772-58-6	5	4	296	COOH	PFHxA
PFHxA	Perfluorohexanoic acid	307-24-4	6	5	314	COOH	
PFHpA	Perfluoroheptanoic acid	375-85-9	7	6	364	COOH	PFOA
PFOA	Perfluorooctanoic acid	335-67-1	8	7	414	COOH	
PFNA	Perfluorononanoic acid	375-95-1	9	8	464	COOH	PFOA
PFDA	Perfluorodecanoic acid	335-76-2	10	9	514	COOH	PFOA
PFUnA	Perfluoroundecanoic acid	2058-94-8	11	10	564	COOH	PFOA
PFDoA	Perfluorododecanoic acid	307-55-1	12	11	614	COOH	PFOA
PFTrDA	Perfluorotridecanonic acid	72629-94-8	13	12	664	COOH	PFOA
PFTDA or PFTA or PFTeDA	Perfluorotetradecanoic acid	376-06-7	14	13	714	COOH	PFOA
PFBS	Perfluorobutane sulfonate	375-73-5	4	4	300	SO3H	
PFEESA	Perfluoro(2-ethoxyethane)sulfonic acid	113507-82-7	4	4	316	SO3H	PFBS
PFPeS	Perfluoropentanesulfonic acid	2706-91-4	5	5	350	SO3H	PFHxS
PFHxS	Perfluorohexane sulfonate	355-46-4	6	6	400	SO3H	
PFHpS	Perfluoroheptanesulfonic acid	375-92-8	7	7	450	SO3H	PFOS
PFOS	Perfluorooctane sulfonate	1763-23-1	8	8	500	SO3H	
9Cl-PF3ONS	9-chlorohexadecafluoro-3-oxanone-1-sulfonic acid	756426-58-1	8	8	533	SO3H	6:2 FTS
PFNS	Perfluorononanesulfonic acid	68259-12-1	9	9	550	SO3H	PFOS
PFDS	Perfluorodecane sulfonic acid	335-77-3	10	10	600	SO3H	PFOS
11Cl-PF3OUdS	11-chloroeicosafluoro-3-oxaundecane-1-sulfonic acid	763051-92-9	10	10	633	SO3H	6:2 FTS
PFDoS or PFDoDS	Perfluorododecanesulfonic acid	79780-39-5	12	12	700	SO3H	PFOS
PFOSA or FOSA	Perfluorooctanesulfonamide	754-91-6	8	8	499	SO2NH2	PFOS
ADONA	4,8-dioxa-3H-perfluorononanoic acid	919005-14-4	7	6	378	COOH	PFOA
HFPO-DA or GenX	Hexafluoropropylene oxide dimer acid	13252-13-6	6	5	330	COOH	PFHxA
4:2 FTS	4:2 Fluorotelomer sulfonate (1H,1H, 2H, 2H-Perfluorohexane sulfonic acid)	757124-72-4	6	4	328	CH2CH2SO3-	6:2 FTS
6:2 FTS	6:2 Fluorotelomer sulfonate (1H,1H, 2H, 2H-Perfluorooctane sulfonic acid)	27619-97-2	8	6	428	CH2CH2SO3-	
8:2 FTS	8:2 Fluorotelomer sulfonate (1H,1H, 2H, 2H-Perfluorodecane sulfonic acid)	39108-34-4	10	8	528	CH2CH2SO3-	6:2 FTS
3:3 FTCA	3:3 Fluorotelomer carboxylic acid (2H,2H,3H,3H-Perfluorohexanoic acid)	356-02-5	6	3	242	CH2COOH	PFBA
5:3 FTCA	5:3 Fluorotelomer carboxylic acid (2H,2H,3H,3H-Perfluorooctanoic acid)	914637-49-3	8	5	342	CH2COOH	6:2 FTS
7:3 FTCA	7:3 Fluorotelomer carboxylic acid (2H,2H,3H,3H-Perfluorodecanoic acid)	812-70-4	10	7	442	CH2COOH	6:2 FTS
MeFOSAA or NMeFOSAA	2-(N-Methylperfluorooctanesulfonamido)acetic acid	2355-31-9	11	8	571	SO2NCCOOH	EtFOSAA
EtFOSAA or NEtFOSAA	N-ethyl perfluorooctanesulfonamidoacetic acid	2991-50-6	12	8	585	SO2NCCOOH	EtFOSAA
MeFOSE or NMeFOSE	N-Methyl perfluorooctane sulfonamidoethanol	24448-09-7	11	8	557	SO2NCH2CH2OH	MeFOSAA
EtFOSE or NEtFOSE	N-Ethyl perfluorooctane sulfonamidoethanol	1691-99-2	12	8	571	SO2NCH2CH3	MeFOSAA
MeFOSA or NMeFOSA	N-Methyl perfluorooctane sulfonamide	31506-32-8	9	8	513	SO2NHCH3	MeFOSAA
EtFOSA or NEtFOSA	N-Ethyl perfluorooctane sulfonamide	4151-50-2	10	8	527	SO2NHCH3	MeFOSAA

Target PFAS highlighted in gray.

Appendix B

PFAS Separation and Destruction Technology Screening Table

Appendix B PFAS Separation and Destruction Technology Screening Table

Technology Category		Technology	Technology description	Highest degree of commercialization (examples: bench-scale, pilot scale, full-scale)	PFAS separation or destruction efficacy	Technology Readiness for PFAS[1]	Municipal WW	Landfill Leachate	Compost Contact Water	Municipal Biosolids	Sorption Media or Concentrates[2]	Demonstrated for PFAS Treatment at Scale [3]?	Demonstrated >90% Efficacy for PFAS[4]?
Separation Technologies	Membrane Separation	Nanofiltration (NF)/Reverse Osmosis (RO) Membrane Separation	PFAS separated into a concentrate stream by physical separation via high-pressure membranes. NF membranes typically have higher water recovery than RO due to larger membrane pore sizes.	Full-scale. Example vendors: Veolia, Evoqua, ROChem.	PFOS separation (membrane rejection) >99% for RO, and 90-99% for NF.[1]	9	X	X	X			Yes	Yes
		Ultrafiltration (UF)	PFAS separated into a concentrate stream by physical separation via low-pressure membranes.	Developing with regard to PFAS separation, though full-scale for other applications.[2]	30-43% PFOS removal, 47-98% PFOA removal. However, UF is not considered effective at removing PFAS because the pore size is larger than most PFAS.[2]	9	X	X	X			Yes	No
	Phase Separation	Foam Fractionation	PFAS stripped from liquid phase as foam using fine air bubbles. This technology takes advantage of the surfactant properties of PFAS at high concentrations. Can use ozone gas instead of air to achieve oxidation of some PFAS (largely PFAA precursors). Often paired with photodegradation or electrochemical oxidation to degrade separated PFAS.	Full-scale with 66,000 gpd application in Australia.[3] Pilot and bench-scale. Example vendors: EPOC Systems (single container units can treat between 50,000-160,000 gpd), Evocra (with ozone).	>96% PFOS removal in AFFF application[4] and 92% PFAS separation in leachate.[5] Up to 80% PFAS removal without ozone, up to 95% PFAS removal bench-scale when combined with ozone, lower removal for short-chain PFAS.[6]	8	X	X	X			Yes	Yes
	Thermal Separation	Thermal Evaporation/ Brine Concentration	Water is evaporated, with most PFAS and other dissolved constituents remaining in a brine or slurry requiring management (for example, dewatering and disposal in a landfill or via high temperature incineration). Some short-chain PFAS may evaporate with water and require downstream treatment. Can be applied with or without vapor condensation or re-compression. Could use waste heat if applied in a landfill setting.[7]	Widely used in wastewater treatment. Example vendor: Veolia.	Very limited data available on PFAS separation and partitioning. Some PFAS, especially smaller molecules, may partition into the air phase and subsequently into condensate.	9	X	X	X		X	Yes	No
		Direct-fired Desorption	Desorb PFAS from a solid media at <650°C where heat source in direct contact with the material of interest, creating a volatilized PFAS gas stream that requires further treatment, typically via thermal oxidation. Commonly applied for soils treatment, activated carbon, and other waste materials with higher throughput than indirect-fired desorption.	Limited information on application to biosolids, though full-scale for other applications. Example vendor: RLC Technologies.	No specific studies found on direct-fired desorption for PFAS in biosolids.	2				X		No	No
		Indirect-fired Desorption	Desorb PFAS from a solid media at <650°C in kiln where heat source is not directly in contact with combusting waste, creating a volatilized PFAS gas stream that requires further treatment, typically via thermal oxidation. Commonly applied for soils treatment and activated carbon.	Pilot-scale.[15]	Can achieve >99.9997% Removal when coupled with thermal oxidation.[8,9]	6				X		No	Yes

Technology Category		Technology	Technology description	Highest degree of commercialization (examples: bench-scale, pilot scale, full-scale)	PFAS separation or destruction efficacy	Technology Readiness for PFAS[1]	Municipal WW	Landfill Leachate	Compost Contact Water	Municipal Biosolids	Sorption Media or Concentrates[2]	Demonstrated for PFAS Treatment at Scale [3]?	Demonstrated >90% Efficacy for PFAS[4]?
Separation Technologies	Biological	Phyto-remediation/ Constructed Wetlands	Contaminants are bioaccumulated in plants. PFAS are not appreciably degraded by plants, but phytoremediation may concentrate PFAS from contaminated soils if plants are removed for disposal/destruction. Flow-through passive treatment using wetland sediments and plants to remove and sequester contaminants from the water phase.	Mature technology at full-scale, limited applications for PFAS treatment.	Limited uptake of PFAS by some plants in the range of 10%, with higher update for short-chain PFAS than longer chain PFAS.[10] Up to 42% of PFPeA recovered in a greenhouse study, the highest recovery of six PFAS studied. The lowest recovery observed for PFOS.[11,12] A mesocosm study of an engineered wetland treatment system for landfill leachate in St. Louis County MN demonstrated PFAS update by bulrush, cattails, and soil medium.[13]	5	X	X	X			Yes	No
	Flocculation/ Coagulation	Alum	PFAS sorbs to or incorporates with particles formed from alum addition and removed via settling with other solids.	Developing with regard to PFAS separation, though full-scale for other applications.[2]	PFOA and PFOS removal ranging from 1-40% has been observed depending on coagulant dose.[2]	7	X	X	X			No	No
		Polyaluminum Chlorides	PFAS sorbs to or incorporates with particles formed from polyaluminum chloride addition and removed via settling with other solids.	Developing with regard to PFAS separation, though full-scale for other applications.[2]	PFOA influent of 50-30,000 µg/L, resulted in 99% removal in one study and PFOA/PFOS influent o 1,000 µg/L, resulted in 1-25% removal based on coagulant dose in another study.[2]	7	X	X	X			No	Yes
		Ferric Salts	PFAS sorbs to or incorporates with particles formed from ferric salt addition and removed via settling with other solids.	Developing with regard to PFAS separation, though full-scale for other applications.[2]	PFOA and PFOS removal ranging from 1-50%. Removal improved to 48-95% when the pH was reduced to 4. Removal is dependent on coagulant dose.[2]	7	X	X	X			No	Yes
		Covalent Bound Hybrid Coagulants	PFAS sorbs to or incorporates with particles formed from addition of covalent-bound hybrid coagulants and removed via settling with other solids.	Bench-scale tests[2]	PFOA influent of 100 ug/L, resulted in 99% removal.[2]	4	X	X	X			No	Yes
		Specialty Coagulants	PFAS sorbs to or incorporates with particles formed from addition of specialty anionic coagulants such as PerfluorAd and removed via settling with other solids.	Developing with regard to PFAS separation, though full-scale for other applications.[2]	PFAS removal ranging from 87-98% on 380-480 ug/L influent. PFOA removal of 20% on 8 ug/L influent and PFOS removal of 80% on 236 ug/L.[2]	7	X	X	X			No	Yes
		Electro-coagulation	PFAS sorbs to or incorporates with particles generated by anode-cathode reactions of metal plates inserted into the water and removed via settling with other solids. The coagulant is created in the process instead of added.	Developing technology.[2]	PFOA removal of 99% on 1,000-100,000 ug/L influent.[2]	5	X	X	X			No	Yes
	Sorption	Granular Activated Carbon (GAC)	PFAS sorbs to hydrophobic GAC surface in a fixed-bed pressure vessel.	Full-scale. Example vendors: Calgon, Cabot, Evoqua.	GAC can remove PFAS to below detection limits until breakthrough occurs. Shorter chain PFAS have lower GAC loading capacities and faster breakthrough times than longer chain PFAS but can be effectively removed with increased change out frequency.[1]	9	X	X	X			Yes	Yes
		Reactivated GAC	Similar to virgin GAC, PFAS sorbs to the hydrophobic GAC surface in a fixed-bed pressure vessel. Reactivated GAC refers specifically to GAC that has been previously used and its sorption capacity has been recovered through thermal reactivation. Reactivated GAC can either be source separated (returned to the same site for reuse) or general pool reactivated GAC (i.e., multiple unknown sources).	Full-scale. Example vendors: Calgon, Cabot, Evoqua	PFAS removed via similar processes as virgin GAC. Similar bed volumes to breakthrough observed for PFOA and PFOS at two different remediation sites with low TOC (2 mg/L).[14] General pool GAC showed approximately 50% faster breakthrough in comparison to virgin and source separated GAC.[15]	9	X	X	X			Yes	Yes
		Carbon Nanotubes and Graphene	PFAS sorbs to the carbon nanotube (CNT) or graphene surface. Often paired with downstream exposure to UV light to photodegrade PFAS.	Developing technology.[2]	CNT and graphene removed PFAS to below detection limits during the sorption process.[16]	4	X	X	X			No	No

Technology Category		Technology	Technology description	Highest degree of commercialization (examples: bench-scale, pilot scale, full-scale)	PFAS separation or destruction efficacy	Technology Readiness for PFAS[1]	Municipal WW	Landfill Leachate	Compost Contact Water	Municipal Biosolids	Sorption Media or Concentrates[2]	Demonstrated for PFAS Treatment at Scale [3]?	Demonstrated >90% Efficacy for PFAS[4]?
Separation Technologies	**Sorption**	Colloidal Activated Carbon	PFAS sorbs to colloidal activated carbon particles in aqueous suspension. Colloidal activated carbon is primarily injected into groundwater to prevent plume mitigation.	Full-scale for remediation, but not for water treatment. Example vendor: Regenesis.	PFOA was reduced by 99% and PFOS by 98% in a shallow silty-sand anaerobic aquifer.[17]	9	X	X	X			No	Yes
		Ion Exchange Resins (Single-Use Media)	PFAS attaches to resin via surface charge interactions in a fixed bed pressure vessel. Once exhausted, media is removed and disposed.	Full-scale, example vendors: Calgon, Evoqua, ECT2, Purolite, ResinTech.	IX can remove PFAS to below detection limits until breakthrough occurs. Recommended for influent concentrations below 10 µg/L total PFAS. Affinity for removal by the resin increases with increasing carbon length within each subgroup of PFAS.[1]	9	X	X	X			Yes	Yes
		Ion Exchange Resins (Regenerable Media)	PFAS attaches to resin via surface charge interactions with resin support material in a fixed-bed pressure vessel. Once exhausted, media is regenerated onsite using a brine/solvent mixture and returned to service.	Full-scale, example vendor: Purolite, ECT2.	IX can remove PFAS to below detection limits until breakthrough occurs. Affinity for removal by the resin increases with increasing carbon length within each subgroup of PFAS. Removal capacity is generally less than single use resin.[1]	9	X	X	X			Yes	Yes
		Biochar	Similar to GAC, PFAS are removed via sorption to biochar in a fixed pressure vessel. The biochar is synthesized by heating biomass. Once exhausted, media is removed and disposed.	Developing technology evaluated at the bench and pilot scale.[2]	PFAS removal is dependent on the biochar type and if it is biologically active or inactive. Inactive biochar can remove 20-60% of short chain PFAS (3-6 carbons) and 90-99% of long chain PFAS [18]	6	X	X	X			No	Yes
		Powdered Activated Carbon (PAC)/Alum/ Kaolinite)	Similar to GAC, PFAS are removed via sorption to composite media of PAC, alum, and kaolinite. The media is added directly in process or tank (not fixed bed). Spent media is wasted and separated by settling or with low-pressure membrane filtration.	Proven technology in *in situ* remediation applications, no known full-scale water treatment applications. Example vendor: AquaBlok.[19]	99.9-99.99% removal of long and short chain PFAS.[20] The media immobilizes PFAS and has a higher affinity for short chain PFAS than GAC.[19]	6	X	X	X			No	Yes
		Super-Fine PAC	PFAS sorbs to powdered activated carbon (PAC) that has been ground to a super-fine powder and added in the process (e.g., within a tank). Superfine PAC then removed via membrane filtration.	Bench-scale tests.[21]	1.5-480 times higher adsorption than GAC, depending on the influent PFAS concentration, adsorbent size, pore content, and PFAS chain length.[21]	4	X	X	X			No	Yes
		Modified Clay	PFAS attaches to clay minerals, sometimes modified, via surface charge interactions. Media is in a fixed bed pressure vessel. Once exhausted, media is removed and disposed.	Full-scale: matCARE, Pilot study: Cetco Fluoro-sorb.	Can remove PFAS to below detection limits until breakthrough occurs. PFOA, PFOS, PFHxS, and PFBS removed to non-detect concentrations, with breakthrough of PFBS observed first.[22]	8	X	X	X			Yes	Yes
		Modified Adsorbents	PFAS sorbs to modified adsorbent media, which can include modified natural materials: polymer-coated sand, modified cyclodextrin, synthesized gel polymeric absorbents, metal-organic frameworks, or modified cellulose.	Developing technology.[2] Example vendor: Cyclopure	Similar or slightly higher PFAS removal to GAC for PFOA and PFOS. PFAS removal depends on the adsorbent modification.[2]	5	X	X	X			No	Yes
	Desorption	Ion Exchange Resin Solvent Regeneration	A proprietary solvent brine solution removes PFAS from the IX media by targeting removal of the ionic head and desorption of the fluorinated carbon tail from the media. The brine containing concentrated PFAS is distilled. Residual still bottoms can be disposed of directly or adsorbed onto a smaller amount of GAC.	Full-scale, example vendor: ECT2.	Regeneration removes sufficient PFAS to restore resin to virgin quality able to handle PFAS loading cycles consistent with virgin resin.[23]	8	X	X	X			Yes	Yes
		GAC Solvent Regeneration	A solvent solution is used to separate PFAS from the GAC media. The brine containing concentrated PFAS is distilled and recycled for future regenerations.	Pilot-scale, ready for commercialization. Example vendor: Battelle.[24]	Resin regeneration can achieve PFAS mass balance recovery of 97%-132%.[24]	6					X	No	Yes

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Destruction Technologies	Redox Chemistry	Solvated Electrons and Advanced Reduction Processes	Solvated electrons are generated in advanced reduction processes (ARP) by combining chemical reductants (e.g., Fe(II), sulfide, sulfite, iodide, or dithionite) with activating energy (e.g., ultrasound, UV, microwaves, or electron beams) to reductively cleave the C-F bonds of PFAS.	Active area of research and development at bench-scale.[1] Example start-up: Enspired Solutions (photo-reductive defluorination).	>90% degradation of PFOA and PFOS has been observed.[25,26] Destruction possible for longer-chain PFAS, but less efficient for short-chain PFAS. Degree of defluorination and reaction rates are dependent on chain length and head group. ARPs can be inhibited by presence of oxidants (e.g., oxygen).[25]	5	X	X	X			No	Yes
		Catalyzed Hydrogen Peroxide (CHP)	Hydrogen peroxide combined with a catalyst such as iron (i.e., for Fenton's reagent) or chelated manganese generate hydroxyl radicals. Certain processes can produce reductants such as superoxide which can mineralize PFAS and convert precursors to terminal PFAS.	Developing with regard to PFAS destruction, though full-scale for other applications.	Hydroxyl radicals attack head group, not fluorinated tail, so limited defluorination occurs. Limited degradation of PFCA and PFSA are observed, with PFAS precursors transformed to terminal PFCAs [1,27]. 68-89% PFOA with CHP with superoxide and hydroperoxide [28] Limited data for PFAS other than PFOA.	4	X	X	X			No	No
		Ozone	Ozone used as an oxidant to degrade contaminants, typically paired with UV light, hydrogen peroxide, or persulfate to generate hydroxyl radicals.	Developing with regard to PFAS destruction, though full-scale for other applications.[1]	Can transform PFAS precursors, but full defluorination unlikely with ozone alone. <25% degradation observed for PFBA, PFPeA, PFOA, PFBS, PFHxS, and PFOS in a bench-scale reactor [29]. Ozone can paired with foam fractionation and/or with UV for improved removal efficiency.[6]	7	X	X	X			No	No
		Activated Persulfate	Persulfate is chemically activated with heat or UV light to form sulfate radicals. Metallic catalyst activation and high pH/alkaline activation (commonly used in remediation applications of activated persulfate) appear to be less effective for PFAS degradation application.[27]	Developing with regard to PFAS destruction, though full-scale for other applications.[1]	Can mineralize PFCAs, but do not readily transform PFSAs.[1] Up to 90% removal of PFOA at bench-scale at 50 deg. C and a long-reaction time, but only 24% defluorination.[30]	7	X	X	X			No	Yes
		Photolysis	Photolysis includes several processes that transform or degrade contaminants using light: • Direct photolysis degrades contaminants that directly absorb light (typically UV light). • UV advanced oxidation processes combine UV light and a chemical oxidant to generate reactive species. • Photocatalysis combines UV light and a solid catalyst (Fe(III) or TiO2) to form reactive species.	Developing with regard to PFAS destruction, though full-scale for other applications.[1]	Limited removal for short-chain PFAS like PFBA and PFBS. 50-90% removal of long-chain with inclusion of a catalyst. Precursors like 6:2 FTS also removed.[31,32] 90% PFAS degradation with carbon nano-tube photocatalysts, with 62% of PFOA and 46% of PFOS completely defluorinated.[16]	7	X	X	X			No	Yes
		Electro-chemical Oxidation (Electrolysis)	Electrical currents passed through water degrade PFAS, either directly on the anode or indirectly in bulk water. May be paired with electrochemical reduction.	Active area of research and development at bench-scale and pilot scale.[1] Example vendor: AECOM De-Fluoro.	15% (PFBA)-97% (PFOA) removal. Preferential removal of PFOA over PFOS because PFOA preferentially adsorbs to anode surface.[33,34]	7	X	X	X		X	No	Yes
		Sonochemical Oxidation	Uses acoustic ultrasound waves (typically 100-1000 kHz) in water to cause cavitation and radical generation. Mechanisms of PFAS degradation are not well-defined in the literature. Additives (e.g., persulfate) or UV light can enhance degradation.	Active area of research and development at bench and pilot scales.[35]	>90% degradation of PFOS has been observed at high frequencies, [36] and can be enhanced with additives or UV light.[37] Some short-chains not fully degraded.[38]	6	X	X	X			No	Yes
		Plasma Technology	A strong electric field is applied to a gas (such as argon), generates ionized gas and electrons, forming strong oxidizing and reducing species that can degrade PFAS, with or without a catalyst.	Active area of research and development at bench and pilot scales. Several SERDP examples.[39]	>90% remove of long-chain PFAAs, 0-95% removal of short-chain PFAS.[40,41]	6	X	X	X			No	Yes

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Destruction Technologies	Redox Chemistry	Zero Valent Iron (ZVI)	Fine-grained metallic iron filings sorb and can reductively degrade contaminants. Mixed with inert sand or gravel in permeable reactive barriers (PRB) for groundwater or surface water treatment.	Developing with regard to PFAS removal/destruction, though full-scale for other applications.[1]	17-95% removal with ZVI for PFOA, PFHpA, PFOS, PFHpS, PFHxS, and PFBS, with 5-10% defluorination of PFOA and PFOS.[42]	7	X	X	X			Yes	No
		Alkaline Metal Reduction	Metallic reductants (e.g., Ti-citrate, nZn0, nNiFe0) reductively defluorinate PFAS, with higher rates under high pH/alkaline conditions.	Not commercially applied. Bench-scale studies have focused on PFOS and isomers.[1]	>95% transformation of PFOS, but intermediates identified; variability in efficiency depending on isomer.[43,44]	4	X	X	X			No	Yes
		Super Critical Water Oxidation (SCWO)	Wastewater, biosolids, or any organic waste (e.g., aqueous film-forming foam, AFFF) subjected to high temperature and pressure (374°C and pressure of 250 bar) to reach supercritical state for water where all organic material dissolve and can be oxidized. To be self-sustaining, waste streams require at least 2-3 MJ/kg or chemical oxygen demand >100 g/L, thus SCWO is applicable for destruction of biosolids and potentially for spent sorption media, but liquid waste streams would likely need a co-fuel.	Developing with regard to PFAS removal/destruction, though full-scale for other applications.[1] Demonstration studies online for biosolids. Example vendors: Battelle PFAS Annihilator, Aquarden Technologies, 374Water.	70% destruction for PFOS.[45] Other studies showed a greater than 99% reduction of the total PFAS identified in a targeted compound analysis, including PFOS and PFOA, with removal efficacy dependent on process design, residence time, and operating parameters.[46,47]	8				X	X	Yes	Yes
		Advanced Oxidation Processes (AOP)	Oxidants such as ozone, peroxide, persulfate, or UV light, produce reactive oxidant radicals to degrade contaminants. Includes previously listed technologies such as photolysis, ozone, and catalyzed hydrogen peroxide.	Multiple AOP are used in full-scale treatment, see specified technologies above for more detail.	See above for specified technologies. Hydroxl radicals attack head group, not fluorinated tail, so limited defluorination occurs. Achieving defluorination typically requires a reduction process as well, so 90% degradation unlikely with just AOP. 75% removal using UV/ozone, but not all defluorinated.[6]	9	X	X	X			Yes	No
		High-Energy Electron Beam (eBeam)	Electricity used to generate electrons, which form solvated electrons and oxidant radicals that facilitate advanced reduction and oxidation without chemical addition.[1]	Commercialized in chemical and food production applications, limited to bench scale for PFAS removal from water or biosolids.[1]	35%-96% defluorination of PFOA, with varying efficiencies based on pH, water chemistry, and oxidant addition,[48,49]	4	X	X	X	X		No	Yes
		Mechano-chemical degradation (MCD)	Solid media treated in high energy ball mills with reactants that produce oxidative radicals and plasma from impact of milling balls. These radicals and plasma react with fluorine to destroy PFAS.	Not developed at commercial scale, but piloted for treatment of contaminated soils.[50]	>99% destruction of PFAS from contaminated soils on lab scale, with limited PFAS byproducts.[50,51]	5				X	X	No	Yes
	Biodegradation	Mycoremediation/ Fungal Enzymes	Using fungi to degrade PFAS. Limited studies to-date. Some studies using lignolytic fungi (wood-rotting fungus).	Active area of research and development at bench scale.[12,52]	Limited data available. 20% degradation of PFOA and PFOS observed.[53] Up to 80% transformation of the precursor 6:2 FTOH degrading to other PFAS.[12,54,55]	3	X	X	X	X		No	No
		Bacterial Enzymes	Enzymes extracted from bacteria used to degrade contaminants. Could include monooxygenases and reductase enzymes used commercially in other applications.	Active area of research and development at bench scale.[56]	Bacterial enzymes that are known to definitively degrade/defluorinate are not well described in the literature.[12]	3	X	X	X	X		No	No
		Biotrans-formation/ defluorination	PFAS are (partially) degraded via microbial degradation under aerobic or anaerobic conditions. PFAS precursors that include non-fluorinated alkyl carbons can be readily transformed into short-chain terminal PFCAs.	Active area of research and development at bench scale.[1,12,56]	Rates of biodegradation are slow. Up to 60% removal of PFOA/PFOS via anaerobic defluorination over 60-day incubation.[57]	4	X	X	X	X		No	No

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Destruction Technologies	Thermal Destruction	GAC Thermal Reactivation	Thermal reactivation occurs in a multiple hearth or rotary kiln furnaces typically operated at approximately 980°C under low oxygen and introduction of steam. PFAS are desorbed and decomposed in the furnace and destroyed via thermal oxidation in an after burner (operated at a higher temperature than the furnace). The process restores most of the sorptive capacity of the media.	Full-scale. Example vendors: Calgon, Evoqua.	PFAS separation during reactivation is PFAS and temperature specific. Defluorination of 80-100% of both PFCAs and PFSAs occur at 1000°C. Decomposition of 80-100% of PFCAs and PFSAs occur at 500°C.[58]	9					x	Yes	Yes
		Hydrothermal Alkaline Treatment (HALT)	Heat and pressure are applied to a reactor at near-supercritical conditions (e.g., 350°C and 165 bar) and alkaline pH (using sodium hydroxide). Under these conditions, water behaves like a nonpolar solvent and shows catalytic degradation of organics. This is a similar process as Hydrothermal Liquefaction for biosolids.	Active area of research and development with prototypes available. Example startup: Aquagga	All 19 targeted PFAS in diluted AFFF samples were degraded in laboratory-scale batch reactor experiments to non-detectable concentrations (which were 100-500 ng/L) within 30 min.[59] PFSAs were more recalcitrant than PFCAs. This study also demonstrated nearly 100% defluorination efficiency.	7	X	X	X			No	Yes
		Low Temperature Mineralization in DMSO	Heat (20-120°C), sodium hydroxide, and the solvent dimethyl sulfoxide (DMSO) in water are used to mineralize PFCAs. DMSO promotes a low energy barrier to decarboxylation (removal of the PFCA head group) and sodium hydroxide promotes degradation to fluoride, small carbon-containing ions (formate, carbonate, oxalate, and glycolate), and trifluoroacetate.	Active area of research at bench scale.[60]	The study by Trang et al. (2022) demonstrated that most PFOA was degraded by over 90% after 24 hours of reaction at 120°C. Starting concentrations were relatively high, at approximately 90 mmol/L (or 37 g/L). Approximately 90% of the fluorine from PFOA was recovered as fluoride, with most of the remainder associated with trifluoroacetic acid. Similar degradation and recoveries were observed for other PFCAs and HFPO-DA. The proposed mechanism is not compatible with PFSAs.	3	X	X	X			No	Yes
		Pyrolysis with Thermal Oxidation (biosolids)	Thermochemical process where pre-dried (>75% total solids) biosolids are heated in a reactor typically from 200-590°C under a starved to no oxygen condition, producing PFAS-laden pyrogas, tars, oils and depending on the operating temperature, either inert material or biochar. Pyrogas routed to thermal oxidation at over 1000°C under excess air/oxygen to destroy PFAS.	Full-scale demonstration projects in place. Example vendors: Anaergia, Bioforcetech, Biowaste.	Complete removal from biochar. Additional studies required for air, water (from air treatment), however with a thermal oxidation PFAS destruction anticipated per below.[61] Temperature greater than 1,100°C for defluorination of PFAS in solids, but flue gas may contain some PFAS. Thermal oxidizer destroys PFAS at an efficiency greater than 99.99%.[8,62]	8				X		Yes	Yes
		Gasification with Thermal Oxidation (biosolids)	Thermochemical process where pre-dried (>75% TS) biosolids are heated in a reactor typically from 590-980°C with limited oxygen (~30% of required O2 for complete oxidation), producing PFAS-laden syngas and biochar. Syngas routed to thermal oxidation at over 1000°C under excess air/oxygen to destroy PFAS.	Full-scale demonstration projects in place. Example vendors: Aries, Ecoremedy.	Limited removal on gasification alone, but 99.5% total PFAS removal between feed and char when coupled with thermal oxidation.[63] Confidential results of full-scale system showed removal from media and air phases. Temperature greater than 1,100°C for defluorination of PFAS in solids, but flue gas may contain some PFAS. Thermal Oxidizer destroys PFAS at an efficiency greater than 99.99%.[8,62]	8				X		Yes	Yes
		Biosolids Incineration	Sewage sludge incinerators (SSIs) operate at 700-1,000°C to further reduce the volume of biosolid residue requiring final disposal. Byproducts are flue gas and ash. There are two types of incineration: multiple hearth furnace and fluidized bed.	Mature technology, full scale.	Limited information, with research ongoing. Some PFAS destruction may occur through an SSI with additional removal through the air pollution control systems. Complete destruction of PFAS (to CO$_2$, H$_2$O, HF, or SO$_2$) requires temperatures over 1,000°C.[64,65]	9				X	X	Yes	No
		Thermal Drying	Thermal dryers use direct or indirect heat to evaporate water from biosolids to reduce the volume of material. Biosolids particles are heated to less than 650°C within the dryer. Steam, particles, and air pollutants from drying process are treated prior to release to the atmosphere.	Mature technology, full scale.	Thermal drying is shown to impact some PFAS precursors, leading to an increase in concentration PFCAs and PFHxA. No evidence that thermal drying destroys PFAS in biosolids. [66]	9				X	X	No	No

Technology Category		Technology	Technology description	Highest degree of commercialization (examples: bench-scale, pilot scale, full-scale)	PFAS separation or destruction efficacy	Technology Readiness for PFAS[1]	Municipal WW	Landfill Leachate	Compost Contact Water	Municipal Biosolids	Sorption Media or Concentrates[2]	Demonstrated for PFAS Treatment at Scale [3]?	Demonstrated >90% Efficacy for PFAS[4]?
Destruction Technologies	Thermal Destruction	High-Temperature Incineration	Thermochemical process heating to greater than 1000°C under excess air/oxygen. This process is commonly used to destroy PFAS present in sorption media following adsorptive treatment.	Full-scale. Example vendors: Veolia, Clean Harbors.	Temperature greater than 1,100°C for defluorination of PFAS in solid materials, but flue gas may contain some PFAS. Thermal Oxidizer controls PFAS at an efficiency greater than 99.99%.[8,62]	9					X	Yes	Yes
		Cement Kiln (biosolids)	Rotary kiln process where biosolids are used as a fuel source to reduce the amount of other fuel (e.g., coal) needed. The biosolids must be dried to >90% to be used in this type of application.	Limited application for PFAS. Should not use more than 20% biosolids to coal fuel to limit air emission effects.[67]	Defluorination over 1,650°C. Thermal degradation currently being demonstrated for PFOA and PFOS in cement kilns by the formation of CaF_2 captured in the cement, using calcium as a catalyst and fluoride as a scavenger.[68] Cement kilns temperatures (1400°C-2000°C) allows for full destruction of PFAS compounds,[67] but major uncertainties remain around temperature and residence time requirements.	3				X	X	Yes	No
		Hydrothermal Liquefaction (HTL)	Uses temperature and pressure to convert wet organic matter, including PFAS-laden biosolids, to biocrude oil and methane rich gas at 360 deg. C, and 3000 psi. The process captures energy to keep the system running.	Demonstration studies only. Example vendor: Genifuel pilot is expected at Metro Vancouver around 2024.	One study found >99% transformation of fluorinated carboxylic acid structures, but lower for sulfonic acid structures.[69] Other findings indicated that HTL under the studied condition was not able to degrade all PFAS existed in sewage sludge.[70]	6				x		No	No

[1] Technology readiness level (TRL) assessment based on criteria outlined by the U.S. DOE in 2010, with TRL 9 reflecting the highest level, with commercial installations operated under the full range of expected conditions[71]. This study required TRL 8 or 9 to pass initial screening

[2] Sorption media refers to PFAS separation or destruction from sorption media such as granular activated carbon or ion exchange media.

[3] The demonstrated at scale criteria requires that liquid treatment technologies have been field implemented and are commercially available and that biosolids treatment technologies have been applied at pilot-scale applications. The different standards are due to biosolids PFAS treatment being less developed than liquid-phase PFAS treatment.

[4] The demonstrated efficacy criteria requires at least 90% PFAS separation or destruction for at least one of the targeted PFAS, which include PFOA, PFOS, PFHxA, PFBA, and PFBS as well as selected other compounds for specific media.

References

1. ITRC. PFAS Treatment Technologies. *PFAS Technical and Regulatory Guidance Document and Fact Sheets* https://pfas-1.itrcweb.org/12-treatment-technologies/ (2022).
2. ITRC. Table 12-1. LIQUID TECHNOLOGIES—REMEDIATION TECHNOLOGIES AND METHODS COMPARISON. *Interstate Technology Regulatory Council: PFAS Technical and Regulatory Guidance Document and Fact Sheets PFAS-1* https://pfas-1.itrcweb.org/ (2022).
3. Australian Government Defence. PFAS Investigation and Management Program: Army Aviation Centre Oakey. https://defence.gov.au/environment/pfas/oakey/managementactivities.asp.
4. Meng, P. *et al.* Efficient removal of perfluorooctane sulfonate from aqueous film-forming foam solution by aeration-foam collection. *Chemosphere* **203**, 263–270 (2018).
5. Robey, N. M., da Silva, B. F., Annable, M. D., Townsend, T. G. & Bowden, J. A. Concentrating Per- and Polyfluoroalkyl Substances (PFAS) in Municipal Solid Waste Landfill Leachate Using Foam Separation. *Environ. Sci. Technol.* **54**, 12550–12559 (2020).
6. Dai, X., Xie, Z., Dorian, B., Gray, S. & Zhang, J. Comparative study of PFAS treatment by UV, UV/ozone, and fractionations with air and ozonated air. *Environ. Sci. Water Res. Technol.* **5**, 1897–1907 (2019).
7. Brown & Caldwell. *Conceptual Leachate Treatment Scoping Study for New England Waste Services of Vermont (NEWSVT) Landfill*. 175 https://legislature.vermont.gov/Documents/2020/WorkGroups/House%20Natural/Contaminants/W~Matt%20Chapman~Conceptual%20Leachate%20Treatment%20Scoping%20Study~1-16-2020.pdf (2019).
8. Barranco, F., Caprio, P. & Hay, G. *Evaluation of Indirect Thermal Desorption Coupled with thermal Oxidation Technology to Treat Solid PFAS-Impacted Investigation-Derived Waste*. 669 https://www.serdp-estcp.org/Program-Areas/Environmental-Restoration/ER18-1572 (2020).

9. Sörengård, M., Lindh, A.-S. & Ahrens, L. Thermal desorption as a high removal remediation technique for soils contaminated with per- and polyfluoroalkyl substances (PFASs). *PLOS ONE* **15**, e0234476 (2020).

10. Zhu, J. *et al.* Juncus sarophorus, a native Australian species, tolerates and accumulates PFOS, PFOA and PFHxS in a glasshouse experiment. *Sci. Total Environ.* **826**, 154184 (2022).

11. Huff, D. K., Morris, L. A., Sutter, L., Costanza, J. & Pennell, K. D. Accumulation of six PFAS compounds by woody and herbaceous plants: potential for phytoextraction. *Int. J. Phytoremediation* **22**, 1538–1550 (2020).

12. Shahsavari, E. *et al.* Challenges and Current Status of the Biological Treatment of PFAS-Contaminated Soils. *Front. Bioeng. Biotechnol.* **8**, 1–15 (2021).

13. Tedrow, O., McQueen, A., Lee, P., Olson, B. & Novak, J. Use of a Pilot-Scale Hybrid Engineered Wetland Treatment System for Removal of Per- / Poly- Fluoroalkyl Substances (PFAS) and Other Constituents of Concern from a Landfill Leachate. in *IWC Proceedings 2022* (2022).

14. Rodowa, A. E. *et al.* Pilot scale removal of per- and polyfluoroalkyl substances and precursors from AFFF-impacted groundwater by granular activated carbon. *Env. Sci Water Res Technol* **6**, 1083–1094 (2020).

15. Westreich, P., Mimna, R., Brewer, J. & Forrester, F. The removal of short-chain and long-chain perfluoroalkyl acids and sulfonates via granular activated carbons: A comparative column study. *Remediat. J.* **29**, 19–26 (2018).

16. Zhao, D. *Cost-effective Destruction of Per- and Polyfluoroalkyl Substances from DoD Subsurface Investigation-derived Wastes using a New Class of Adsorptive Photocatalysts*. https://www.serdp-estcp.org/Program-Areas/Environmental-Restoration/ER18-1515 (2020).

17. McGregor, R. In Situ treatment of PFAS-impacted groundwater using colloidal activated Carbon. *Remediat. J.* **28**, 33–41 (2018).

18. Dalahmeh, S. S., Alziq, N. & Ahrens, L. Potential of biochar filters for onsite wastewater treatment: Effects of active and inactive biofilms on adsorption of per- and polyfluoroalkyl substances in laboratory column experiments. *Environ. Pollut.* **247**, 155–164 (2019).
19. AquBlok. Rembind Product Overview: Immobilizing Soil Contaminants. (2018).
20. Riegel, M., Enger, S. & Sacher, F. *Review of water treatment systems for PFAS removal*. https://www.concawe.eu/wp-content/uploads/Rpt_20-14.pdf (2020).
21. Murray, C. C. *et al.* Removal of per- and polyfluoroalkyl substances using super-fine powder activated carbon and ceramic membrane filtration. *J. Hazard. Mater.* **366**, 160–168 (2019).
22. Pannu, M. & Plumlee, M. *PFAS Phase I Pilot-Scale Treatment Study Final Report*. https://www.ocwd.com/media/9829/2021-03-24_ocwd-pfas-pilot-i_finalreport.pdf (2021).
23. Woodard, S., Berry, J. & Newman, B. Ion exchange resin for PFAS removal and pilot test comparison to GAC. *Remediat. J.* **27**, 19–27 (2017).
24. Longbrake, D. Battelle's GAC Regeneration Technology. (2022).
25. Cui, J., Gao, P. & Deng, Y. Destruction of Per- and Polyfluoroalkyl Substances (PFAS) with Advanced Reduction Processes (ARPs): A Critical Review. *Environ. Sci. Technol.* **54**, 3752–3766 (2020).
26. Gu, Y., Dong, W., Luo, C. & Liu, T. Efficient Reductive Decomposition of Perfluorooctanesulfonate in a High Photon Flux UV/Sulfite System. *Environ. Sci. Technol.* **50**, 10554–10561 (2016).
27. Dombrowski, P. M. *et al.* Technology review and evaluation of different chemical oxidation conditions on treatability of PFAS. *Remediat. J.* **28**, 135–150 (2018).
28. Mitchell, S. M., Ahmad, M., Teel, A. L. & Watts, R. J. Degradation of Perfluorooctanoic Acid by Reactive Species Generated through Catalyzed H2O2 Propagation Reactions. *Environ. Sci. Technol. Lett.* **1**, 117–121 (2014).

29. Franke, V., Schäfers, M. D., Joos Lindberg, J. & Ahrens, L. Removal of per- and polyfluoroalkyl substances (PFASs) from tap water using heterogeneously catalyzed ozonation. *Env. Sci Water Res Technol* **5**, 1887–1896 (2019).
30. Yin, P., Hu, Z., Song, X., Liu, J. & Lin, N. Activated Persulfate Oxidation of Perfluorooctanoic Acid (PFOA) in Groundwater under Acidic Conditions. *Int. J. Environ. Res. Public. Health* **13**, 602 (2016).
31. Chowdhury, N., Prabakar, S. & Choi, H. Dependency of the photocatalytic and photochemical decomposition of per- and polyfluoroalkyl substances (PFAS) on their chain lengths, functional groups, and structural properties. *Water Sci. Technol.* **84**, 3738–3754 (2021).
32. Olatunde, O. C., Kuvarega, A. T. & Onwudiwe, D. C. Photo enhanced degradation of polyfluoroalkyl and perfluoroalkyl substances. *Heliyon* **6**, e05614 (2020).
33. Radjenovic, J., Duinslaeger, N., Avval, S. S. & Chaplin, B. P. Facing the Challenge of Poly- and Perfluoroalkyl Substances in Water: Is Electrochemical Oxidation the Answer? *Environ. Sci. Technol.* **54**, 14815–14829 (2020).
34. U.S. EPA. *Potential PFAS Destruction Technology: Electrochemical Oxidation*. https://www.epa.gov/sites/default/files/2021-01/documents/pitt_research_brief_electrochemical_oxidation_final_jan_25_2021_508.pdf (2021).
35. Divine, C. Sonolysis-Based In Situ PFAS Treatment Within an HRX Well (ER21-5045). https://serdp-estcp.org/index.php//Program-Areas/Environmental-Restoration/ER21-5045 (2021).
36. James Wood, R. *et al.* Ultrasonic degradation of perfluorooctane sulfonic acid (PFOS) correlated with sonochemical and sonoluminescence characterisation. *Ultrason. Sonochem.* **68**, 105196 (2020).
37. Cao, H., Zhang, W., Wang, C. & Liang, Y. Sonochemical degradation of poly- and perfluoroalkyl substances – A review. *Ultrason. Sonochem.* **69**, 105245 (2020).

38. Sidnell, T., James Wood, R., Hurst, J., Lee, J. & Bussemaker, M. J. Sonolysis of Per- And Poly Fluoroalkyl Substances (PFAS): A meta-analysis. *Ultrason. Sonochem.* 105944 (2022) doi:10.1016/j.ultsonch.2022.105944.

39. Hagelin, N. Removal and Destruction of PFAS and Co-Occurring Chemicals from Groundwater via Extraction and Treatment with Ion Exchange Media, and On-Site Regeneration, Distillation, and Plasma Destruction (ER18-5015). https://www.serdp-estcp.org/Program-Areas/Environmental-Restoration/ER18-5015 (2018).

40. Nau-Hix, C. *et al.* Field Demonstration of a Pilot-Scale Plasma Reactor for the Rapid Removal of Poly- and Perfluoroalkyl Substances in Groundwater. *ACS EST Water* **1**, 680–687 (2021).

41. Singh, R. K. *et al.* Removal of Poly- and Per-Fluorinated Compounds from Ion Exchange Regenerat Still Bottom Samples in a Plasma Reactor. *Environ. Sci. Technol.* **54**, 13973–13980 (2020).

42. Liu, Y., Ptacek, C. J., Baldwin, R. J., Cooper, J. M. & Blowes, D. W. Application of zero-valent iron coupled with biochar for removal of perfluoroalkyl carboxylic and sulfonic acids from water under ambient environmental conditions. *Sci. Total Environ.* **719**, 137372 (2020).

43. Park, S., de Perre, C. & Lee, L. S. Alternate Reductants with VB12 to Transform C8 and C6 Perfluoroalkyl Sulfonates: Limitations and Insights into Isomer-Specific Transformation Rates, Products and Pathways. *Environ. Sci. Technol.* **51**, 13869–13877 (2017).

44. Zenobio, J. E., Modiri-Gharehveran, M., de Perre, C., Vecitis, C. D. & Lee, L. S. Reductive transformation of perfluorooctanesulfonate by nNiFe0-Activated carbon. *J. Hazard. Mater.* **397**, 122782 (2020).

45. Pinkard, B. R., Shetty, S., Stritzinger, D., Bellona, C. & Novosselov, I. V. Destruction of perfluorooctanesulfonate (PFOS) in a batch supercritical water oxidation reactor. *Chemosphere* **279**, 130834 (2021).

46. Krause, M. J. *et al.* Supercritical Water Oxidation as an Innovative Technology for PFAS Destruction. *J. Environ. Eng.* **148**, 05021006 (2022).
47. Li, J., Austin, C., Moore, S., Pinkard, B. R. & Novosselov, I. V. PFOS destruction in a continuous supercritical water oxidation reactor. *Chem. Eng. J.* **451**, 139063 (2023).
48. Kim, T.-H. *et al.* Decomposition of perfluorooctane sulfonate (PFOS) using a hybrid process with electron beam and chemical oxidants. *Chem. Eng. J.* **361**, 1363–1370 (2019).
49. Wang, L., Batchelor, B., Pillai, S. D. & Botlaguduru, V. S. V. Electron beam treatment for potable water reuse: Removal of bromate and perfluorooctanoic acid. *Chem. Eng. J.* **302**, 58–68 (2016).
50. Berg, C. *et al.* Developing innovative treatment technologies for PFAS-containing wastes. *J. Air Waste Manag. Assoc.* **72**, 540–555 (2022).
51. Turner, L. P. *et al.* Mechanochemical remediation of perfluorooctanesulfonic acid (PFOS) and perfluorooctanoic acid (PFOA) amended sand and aqueous film-forming foam (AFFF) impacted soil by planetary ball milling. *Sci. Total Environ.* **765**, 142722 (2021).
52. Zhang, J. MnDRIVE Environment: Using Fungi to Remediate 'Forever Chemicals'. https://mndrive.umn.edu/environment/using-fungus-remediate-forever-chemicals (2021).
53. Tseng, N. S. Feasibility of Biodegradation of Polyfluoroalkyl and Perfluoroalkyl Substances. (University of California Los Angeles, 2012).
54. Merino, N. *et al.* Fungal biotransformation of 6:2 fluorotelomer alcohol. *Remediat. J.* **28**, 59–70 (2018).
55. Tseng, N., Wang, N., Szostek, B. & Mahendra, S. Biotransformation of 6:2 Fluorotelomer Alcohol (6:2 FTOH) by a Wood-Rotting Fungus. *Environ. Sci. Technol.* **48**, 4012–4020 (2014).
56. Wackett, L. P. Nothing lasts forever: understanding microbial biodegradation of polyfluorinated compounds and perfluorinated alkyl substances. *Microb. Biotechnol.* **15**, 773–792 (2022).

57. Huang, S. & Jaffé, P. R. Defluorination of Perfluorooctanoic Acid (PFOA) and Perfluorooctane Sulfonate (PFOS) by *Acidimicrobium* sp. Strain A6. *Environ. Sci. Technol.* **53**, 11410–11419 (2019).
58. Sonmez Baghirzade, B. *et al.* Thermal Regeneration of Spent Granular Activated Carbon Presents an Opportunity to Break the Forever PFAS Cycle. *Environ. Sci. Technol.* **55**, 5608–5619 (2021).
59. Hao, S. *et al.* Hydrothermal Alkaline Treatment for Destruction of Per- and Polyfluoroalkyl Substances in Aqueous Film-Forming Foam. *Environ. Sci. Technol.* **55**, 3283–3295 (2021).
60. Trang, B. *et al.* Low-temperature mineralization of perfluorocarboxylic acids. *Science* **377**, 839–845 (2022).
61. Thoma, E. D. *et al.* Pyrolysis processing of PFAS-impacted biosolids, a pilot study. *J. Air Waste Manag. Assoc. 1995* **72**, 309–318 (2022).
62. Focus Environmental Inc. Thermal Oxidizer Performance Test Report: Chemours Company Fayetteville Works. 65 (2020).
63. Davis, B. *Application of Aries Process Technology to the Problem of PFAS Contamination*. 1–7 https://ariescleantech.com/white-paper-pfas-application-of-aries-process-technology-to-the-problem-of-pfas-contamination/ (2020).
64. Winchell, L. J. *et al.* Per- and polyfluoroalkyl substances thermal destruction at water resource recovery facilities: A state of the science review. *Water Environ. Res.* **93**, 826–843 (2021).
65. Winchell, L. *et al.* Fate of PFAS through Incineration at Wastewater Reclamation Facilities. in (2022).
66. Lazcano, R. K., de Perre, C., Mashtare, M. L. & Lee, L. S. Per- and poly-fluoroalkyl substances in commercially available biosolid-based fertilizers: the effect of post-treatment processes. in (2019).
67. Patterson, C. & Dastgheib, S. A. Cement Kiln and Waste to Energy Incineration of Spent Media. (2020).
68. Wahlstrom, M. & Pohjalainen, E. *Fluorinated polymers in a low carbon, circular and toxic-free economy*. (2021).

69. Yu, J., Nickerson, A., Li, Y., Fang, Y. & Strathmann, T. J. Fate of per- and polyfluoroalkyl substances (PFAS) during hydrothermal liquefaction of municipal wastewater treatment sludge. *Environ. Sci. Water Res. Technol.* **6**, 1388–1399 (2020).

70. Zhang, W. & Liang, Y. Hydrothermal liquefaction of sewage sludge – effect of four reagents on relevant parameters related to biocrude and PFAS. *J. Environ. Chem. Eng.* **10**, 107092 (2022).

71. US DOE. *Technology Readiness Assessment (TRA)/Technology Maturation Plan (TMP) Process Guide*. https://www.energy.gov/sites/prod/files/em/Volume_I/O_SRP.pdf (2010).

Appendix C

PFAS Removal Performance by Technology

Technology	Reference	Study Description	PFBA	PFBS	PFHxA	PFHxS	PFOA	PFOS	6:2 FTSA	N-EtFOSAA	N-MeFOSAA
Separation – Membrane Separation		**RO/NF: Rejection (%) = $(1 - PFAS_{Permeate}/PFAS_{Feed}) \times 100\%$**									
RO/NF	Appleman et al. (2013)	Study evaluating polyamide membrane NF270 for the treatment of PFAS-spiked water. Three experiments were performed: virgin membranes fed with spiked DI water, and spiked artificial groundwater, and fouled membranes fed with spiked artificial groundwater. Results presented as range over various pressure and membrane/water configurations. Results from Table 2.	93-95	95-99	95	96-99	95-97	99	--	--	--
	Appleman et al. (2014)	Study evaluating 15 full-scale treatment systems for the treatment of PFAS compounds. Source waters for treatment systems included 11 surface waters, 6 groundwaters, 1 surface water and groundwater blend, and 2 treated wastewaters. Results from Table 2.	82-95	93-98	97-99	90-96	47-98	96-99	--	55-58	36-84
	Glover et al. (2018)	Full- and pilot-scale potable reuse plants. Average rejection rates are from Table 2. Rejection rates are constrained by analytical reporting limits.	>79	--	>96	>80	>77	>71	>78	--	--
	Franke et al. (2019)	Pilot-scale NF treatment system in Sweden treating groundwater. Rejection was calculated from Table 2.	98	95.7	95.4	95.6	96.5	95	--	--	--
Separation – Phase Separation		**Foam Fractionation: Removal Efficiency (%) = $(1 - PFAS_{Effluent}/PFAS_{Influent}) \times 100\%$**									
Foam Fractionation	Robey et al. (2020)	Bench-scale testing of landfill leachate. Removal estimates from Figures 2 and 3, rounded to the nearest 5%.	-5	25	50	95	95	95	70	95	95
	McCleaf et al. (2021)	Municipal landfill leachate treated in batch column tests. Removal estimates from Table 3.	38	60	66	97	99	98	90	40	11
	Smith et al. (2022)	Pilot-scale treatment of landfill leachate using a continuous system. Removal estimates from Figure SI.2, rounded to the nearest 5%.	0	10	25	90	90	95	90	90	90
	Burns et al. (2022)	Commercial-scale treatment of landfill leachate from Telge Recycling plant in Sweden. Removal estimates from Tables 5 and 6. Removal efficiency estimates were constrained by analytical reporting limits.	-1.1	15.7	37.3	>98.8	>99.7	>98.7	>97.2	97.3	>64.3
Separation – Media Sorption		**Approximate Bed Volumes to First Detection of Breakthrough**									
GAC	Woodard et al. (2017)	Pilot-scale study using Calgon F400 GAC to treat a groundwater matrix with four vessels in series with an overall EBCT of 20 minutes. Approximate bed volumes to breakthrough estimated from Exhibits 11-21.	~3,000	~3,500	~3,000	~3,000	~3,000	~3,000	~3,000	--	--
	Westreich et al. (2018)	This study using rapid small-scale column tests (RSSCT) to simulate a large-scale, fixed GAC bed. The tests were run for ~100,000 bed volumes. The tests were conducted using Calgon F400 and two types of reactivated carbon and untreated groundwater spiked with PFAS. Approximate bed volumes to breakthrough shown were estimated from Exhibits 4 through 10.	~10,000	~65,000–75,000	~55,000–70,000	~85,000–100,000	~75,000–85,000	~95,000–100,000	--	--	--
	Franke et al. (2019)	Full-scale nanofiltration membrane pilot plant combined with column adsorption experiments applying GAC and AIX material. Two different water types were tested, untreated groundwater and membrane reject water. AIX (A600) and GAC (F400). Approximate bed volumes to breakthrough estimated from Figure 2.	--	~5,000	~5,000	~5,000	~5,000	~5,000	--	--	--
	Liu et al. (2021)	Pilot scale system fed with groundwater compared four granular activated carbons The system was run with an empty bed contact time (EBCT) of 10 minutes per vessel. Approximate bed volumes to breakthrough estimated from Figure 1 for Calgon F400 GAC.	--	~15,000	~14,000	~17,500	~19,000	~27,000	--	--	--
AIX	Woodard et al. (2017)	Used ECT's Sorbix A3F to treat a groundwater matrix using three resin vessels in series with an overall EBCT of 7.5 minutes. Pilot test included an initial loading cycle to evaluate virgin resin removal capacity and subsequent alternating cycles of resin loading and regeneration to evaluate effectiveness of regenerated resin. Approximate bed volumes to breakthrough estimated from Exhibits 11-21.	~4,000	~21,000	~10,000	~21,000	~13,000	~21,000	~7,000	--	--
	Franke et al. (2019)	Full-scale nanofiltration membrane pilot plant combined with column adsorption experiments applying GAC and AIX material. Two different water types were tested, untreated groundwater and membrane reject water. AIX (A600) and GAC (F400). Approximate bed volumes to breakthrough estimated from Figure 2.	--	~18,000	~8,000	~17,000	~17,000	~19,000	--	--	--
	Boyer et al. (2021)	Review article summarizing use of anion exchange resins for PFAS removal. Table 3 of this review article summarizes bed volumes to breakthrough for various resins, media, and PFAS.	~698–120,000	~9,000–1,100,000	~2,764–70,000	~15,000–1,100,00	~100–1,100,000	~450–1,100,000	~10,000–120,500	--	--
	Ellis et al. (2022)	Pilot study using groundwater contaminated with aqueous film-forming foam. A series of resins were compared: A520E, PFA694E, CalRes 2301, and PSR2+ (A860 was also tested, BV to breakthrough are not shown here). Approximate bed volumes to breakthrough estimated from Figures 2 and Table 3.	~5,000–20,000	~70,000–180,000	~10,000–70,000	~120,000–180,000	~30,000–180,000	~180,000	--	--	--

Technology	Reference	Study Description	PFBA	PFBS	PFHxA	PFHxS	PFOA	PFOS	6:2 FTSA	N-EtFOSAA	N-MeFOSAA
Modified Clay	Najm et al. (2021)	RSSCTs compared PFAS removal between three types of GAC and modified clay (Fluoro-sorb 200). Tests were completed using groundwater used for drinking water from San Jose, California. Influent PFBS, PFHxS, and PFOS concentrations were 2.4 ng/L, 8.5 ng/L, and 7.8 ng/L, respectively. The modified clay RSSCT column was operated at a full-scale equivalent of 2-minute EBCT. Note that PFHxA was detected in the influent at just below 2 ng/L (below the analytical reporting limit). PFHxA broke through at 30,000 BV. No explanation was provided for the relatively low capacity of the modified clay for PFHxA.	--	>300,000	30,000 (refer to description)	>300,000	--	>300,000	--	--	--
	Grieco et al. (2021)	RSSCTs compared PFAS removal between GAC (F400) and modified clay (Fluoro-sorb 200). Both media were ground prior to testing. RSSCTs were completed using groundwater used for drinking water from the Orange County Water District. The modified clay RSSCT column was operated at a full-scale equivalent of 2 minute EBCT. Bed volumes shown are RSSCT bed volumes estimated from Figure 5. The range is based on samples with varying dissolved organic carbon concentrations. The influent PFBS and PFOA concentrations ranged from 2.93 to 15.6 ng/L and 6.59 to 22.0 ng/L, respectively.	--	~50,000-175,000	--	--	~225,000 -500,000	--	--	--	--
Separation – Media Sorption		**Removal Efficiency from Batch Isotherm or Kinetic Tests**									
Modified Clay	Yan et al. (2020)	PFAS-impacted groundwater collected from a former US Air Force base was used in batch adsorption experiments. The treatment of PFAS compounds with modified smectite clay (Fluoro-sorb 200) was studied. Removal efficiencies are estimated from Figure 1.	--	--	--	~94	~88	~94	~77	--	--
	Jiang et al. (2022)	Modified clay media was prepared in the laboratory using montmorillonite and the quaternary ammonium surfactant cetyltrimethylammonium chloride (CTAC). A series of batch isotherm and kinetic tests were completed. Five concentrations ranges of PFAS were tested in synthetic laboratory water. PFAS were dosed in combination. Removal efficiencies shown are estimated from Figure 4 for the level 5 PFAS concentrations (highest concentrations tested) after 4 hours of equilibration. Removal efficiencies at lower concentration ranges tested were all >80% at all equilibration times (1-24 hours).	--	>60	>50	>95	>95	~100	>90	~100	--
Destruction – Redox Chemistry		**Destruction (%) = 100 x (1 - ($PFAS_{effluent}$/$PFAS_{influent}$))**									
SCWO	Pinkard et al. (2021)	PFOS decomposition in a batch SCWO reactor. Hydrogen peroxide was used as the oxidant source. Compared to continuous SCWO reactors, batch reactors operate at long residence times and lower temperatures. Results of this study are presented as destruction efficiencies at 425–500°C, 0-min and 60-min residence time. Destruction efficiencies estimated from Figure 2.	--	--	--	--	--	39–70	--	--	--
	Krause et al. (2022)	Article summarizing three demonstration studies done by providers of SCWO systems (Battelle, Aquarden Technologies, and 374Water). PFOS-based AFFF (3M Lightwater) was used as the test solution. All three demonstrations were continuous SCWO systems. Destruction efficiencies estimated from Figure 1.	99–100	99–100	99–100	99–100	99–100	98–99	--	--	--
	McDonough et al. (2022)	A 1,000-fold AFFF foam was used for this pilot-scale study. Destruction efficiencies from Table 4.	99.97	99.99	99.99	99.99	99.99	99.99	--	--	--
Destruction – Thermal Destruction		**Decomposition (%) = (1 – $Mass_{Final}$/$Mass_{Initial}$)×100%**									
GAC Reactivation	Xiao et al. (2020)	Laboratory study of the decomposition of PFAS on the surface of GAC under varying heat treatment temperatures (200-900°C) and atmospheres flow (air, N_2, CO_2). Decomposition efficiencies from Figure 3a-h.	94.31–99.98	97.51–99.97	--	98.3–99.94	70.06–99.97	98.67–100	--	--	--
	Sonmez Baghirzade et al. (2021)	Review article of thermal reactivation of GAC. Percent decomposition ranges shown for temperatures between 175°C and 500°C from Figure 2a and Figure 3. Values for N_2 environment. Values estimated to the nearest 5%.	90–95	0–95	--	--	70–95	0–95	--	--	--
	Sasi et al., (2021)	Laboratory study of the effects of GAC and other porous media on PFAS thermal decomposition. Percent decomposition ranges shown for temperatures 150°C and 300°C. No pressure information given assuming atmospheric pressure. Values estimate from logarithmic plots in Figure 1.	0–99	--	--	--	0–99	--	--	--	--
	DiStefano et al. (2022)	GAC reactivation conducted at facility owned and operated by Calgon. PFAS extraction and analysis from spent and reactivated GAC samples during emissions testing. Destruction efficiencies calculated from Table 1 and averaged over the three emissions tests.	~99.97	~99.98	~99.98	~99.99	~99.99	~99.99	~99.13	~99.63	~99.68
Pyrolysis	Barranco et al. (2020)	A series of tests conducted on sand spiked with PFAS ranging from 6,000 to 19,000 µg/kg. Results at 650°C. No pressure information given assuming atmospheric pressure. Destruction efficiency estimated from Figure 4-2.	--	99.98	--	99.98	99.99	99.98	--	--	--

Technology	Reference	Study Description	PFBA	PFBS	PFHxA	PFHxS	PFOA	PFOS	6:2 FTSA	N-EtFOSAA	N-MeFOSAA
	Thoma et al., (2022)	Study of a commercial pyrolysis system. No PFAS were detected in the resulting biochar. Results are reported as a range from the two labs used to test PFAS concentrations. Values from Table 2 in the SI.	98.07–99.78	96.55	99.74–99.95	--	99.75–99.96	99.08–99.66	81.32	98.39–99.37	98.83–99.48

RO=reverse osmosis; NF=nanofiltration; GAC=granular activated carbon; AIX=anion exchange; SCWO=supercritical water oxidation; -- = Data not reported.

References

Appleman, T. D., Dickenson, E. R. V., Bellona, C., & Higgins, C. P. (2013). Nanofiltration and granular activated carbon treatment of perfluoroalkyl acids. *Journal of Hazardous Materials*, *260*, 740–746. https://doi.org/10.1016/j.jhazmat.2013.06.033

Appleman, T. D., Higgins, C. P., Quiñones, O., Vanderford, B. J., Kolstad, C., Zeigler-Holady, J. C., & Dickenson, E. R. V. (2014). Treatment of poly- and perfluoroalkyl substances in U.S. full-scale water treatment systems. *Water Research*, *51*, 246–255. https://doi.org/10.1016/j.watres.2013.10.067

Barranco, F., Caprio, P., & Hay, G. (2020). *Evaluation of Indirect Thermal Desorption Coupled with thermal Oxidation Technology to Treat Solid PFAS-Impacted Investigation-Derived Waste* (SERDP Project ER18-1572; p. 669). Strategic Environmental Research and Development Program (SERDP). https://www.serdp-estcp.org/Program-Areas/Environmental-Restoration/ER18-1572

Boyer, T. H., Fang, Y., Ellis, A., Dietz, R., Choi, Y. J., Schaefer, C. E., Higgins, C. P., & Strathmann, T. J. (2021). Anion exchange resin removal of per- and polyfluoroalkyl substances (PFAS) from impacted water: A critical review. *Water Research*, *200*, 117244. https://doi.org/10.1016/j.watres.2021.117244

Burns, D. J., Hinrichsen, H. M., Stevenson, P., & Murphy, P. J. C. (2022). Commercial-scale remediation of per- and polyfluoroalkyl substances from a landfill leachate catchment using Surface-Active Foam Fractionation (SAFF). *Remediation Journal*, *32*, 139–150.

DiStefano, R., Feliciano, T., Mimna, R. A., Redding, A. M., & Matthis, J. (2022). Thermal destruction of PFAS during full-scale reactivation of PFAS-laden granular activated carbon. *Remediation Journal*, *32*(4), 231–238. https://doi.org/10.1002/rem.21735

Ellis, A. C., Liu, C. J., Fang, Y., Boyer, T. H., Schaefer, C. E., Higgins, C. P., & Strathmann, T. J. (2022). Pilot study comparison of regenerable and emerging single-use anion exchange resins for treatment of groundwater contaminated by per- and polyfluoroalkyl substances (PFASs). *Water Research*, *223*, 119019. https://doi.org/10.1016/j.watres.2022.119019

Franke, V., McCleaf, P., Lindegren, K., & Ahrens, L. (2019). Efficient removal of per- and polyfluoroalkyl substances (PFASs) in drinking water treatment: Nanofiltration combined with active carbon or anion exchange. *Environmental Science: Water Research & Technology*, *5*(11), 1836–1843. https://doi.org/10.1039/C9EW00286C

Glover, C. M., Quiñones, O., & Dickenson, E. R. V. (2018). Removal of perfluoroalkyl and polyfluoroalkyl substances in potable reuse systems. *Water Research*, *144*, 454–461. https://doi.org/10.1016/j.watres.2018.07.018

Grieco, S. A., Chang, J., Maio, E. (Lily) Y., & Hwang, M. (2021). Comparing conventional and emerging adsorbents for per- and polyfluoroalkyl substances: Kinetic, equilibrium, and column experiments. *AWWA Water Science*, *3*(6). https://doi.org/10.1002/aws2.1256

Jiang, T., Zhang, W., Ilango, A. K., Feldblyum, J. I., Wei, Z., Efstathiadis, H., Yigit, M. V., & Liang, Y. (2022). Surfactant-Modified Clay for Adsorption of Mixtures of Per- and Polyfluoroalkyl Substances (PFAS) in Aqueous Solutions. *ACS Applied Engineering Materials*, acsaenm.2c00096. https://doi.org/10.1021/acsaenm.2c00096

Krause, M. J., Thoma, E., Sahle-Damesessie, E., Crone, B., Whitehill, A., Shields, E., & Gullett, B. (2022). Supercritical Water Oxidation as an Innovative Technology for PFAS Destruction. *Journal of Environmental Engineering*, *148*(2), 05021006. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001957

McCleaf, P., Kjellgren, Y., & Ahrens, L. (2021). Foam fractionation removal of multiple per- and polyfluoroalkyl substances from landfill leachate. *AWWA Water Science*, *3*(5), e1238. https://doi.org/10.1002/aws2.1238

McDonough, J. T., Kirby, J., Bellona, C., Quinnan, J. A., Welty, N., Follin, J., & Liberty, K. (2022). Validation of supercritical water oxidation to destroy perfluoroalkyl acids. *Remediation Journal*, *32*(1–2), 75–90. https://doi.org/10.1002/rem.21711

Najm, I., Gallagher, B., Vishwanath, N., Blute, N., Gorzalski, A., Feffer, A., & Richardson, S. (2021). Per- and polyfluoroalkyl substances removal with granular activated carbon and a specialty adsorbent: A case study. *AWWA Water Science*, *3*(5). https://doi.org/10.1002/aws2.1245

Pinkard, B. R., Shetty, S., Stritzinger, D., Bellona, C., & Novosselov, I. V. (2021). Destruction of perfluorooctanesulfonate (PFOS) in a batch supercritical water oxidation reactor. *Chemosphere*, *279*, 130834. https://doi.org/10.1016/j.chemosphere.2021.130834

Robey, N. M., da Silva, B. F., Annable, M. D., Townsend, T. G., & Bowden, J. A. (2020). Concentrating Per- and Polyfluoroalkyl Substances (PFAS) in Municipal Solid Waste Landfill Leachate Using Foam Separation. *Environmental Science & Technology*, *54*(19), 12550–12559. https://doi.org/10.1021/acs.est.0c01266

Sasi, P. C., Alinezhad, A., Yao, B., Kubátová, A., Golovko, S. A., Golovko, M. Y., & Xiao, F. (2021). Effect of granular activated carbon and other porous materials on thermal decomposition of per- and polyfluoroalkyl substances: Mechanisms and implications for water purification. *Water Research*, *200*, 117271. https://doi.org/10.1016/j.watres.2021.117271

Smith, S. J., Wiberg, K., McCleaf, P., & Ahrens, L. (2022). Pilot-Scale Continuous Foam Fractionation for the Removal of Per- and Polyfluoroalkyl Substances (PFAS) from Landfill Leachate. *ACS ES&T Water*, *2*(5), 841–851. https://doi.org/10.1021/acsestwater.2c00032

Sonmez Baghirzade, B., Zhang, Y., Reuther, J. F., Saleh, N. B., Venkatesan, A. K., & Apul, O. G. (2021). Thermal Regeneration of Spent Granular Activated Carbon Presents an Opportunity to Break the Forever PFAS Cycle. *Environmental Science & Technology*, *55*(9), 5608–5619. https://doi.org/10.1021/acs.est.0c08224

Thoma, E. D., Wright, R. S., George, I., Krause, M., Presezzi, D., Villa, V., Preston, W., Deshmukh, P., Kauppi, P., & Zemek, P. G. (2022). Pyrolysis processing of PFAS-impacted biosolids, a pilot study. *Journal of* *the Air & Waste Management Association*, *72*(4), 309–318. https://doi.org/10.1080/10962247.2021.2009935

Westreich, P., Mimna, R., Brewer, J., & Forrester, F. (2018). The removal of short-chain and long-chain perfluoroalkyl acids and sulfonates via granular activated carbons: A comparative column study. *Remediation Journal*, *29*(1), 19–26. https://doi.org/10.1002/rem.21579

Woodard, S., Berry, J., & Newman, B. (2017). Ion exchange resin for PFAS removal and pilot test comparison to GAC. *Remediation Journal*, *27*(3), 19–27. https://doi.org/10.1002/rem.21515

Xiao, F., Sasi, P. C., Yao, B., Kubátová, A., Golovko, S. A., Golovko, M. Y., & Soli, D. (2020). Thermal Stability and Decomposition of Perfluoroalkyl Substances on Spent Granular Activated Carbon. *Environmental Science & Technology Letters*. https://doi.org/10.1021/acs.estlett.0c00114

Yan, B., Munoz, G., Sauve, S., Liu, J. (2020). Molecular mechanisms of per- and polyfluoroalkyl substances on a modified clay: a combined experimental and molecular simulation study. Journal of Water Research, 184. https://doi.org/10.1016/j.watres.2020.116166 .

Appendix D

Media Breakthrough Curve Estimates

Appendix D: Media Breakthrough Curve Estimates

1 Introduction

Breakthrough modeling was completed to estimate the time (and volume of water treated) before target per- and polyfluoroalkyl substances (PFAS) breakthrough fixed bed media vessels to effluent concentrations matching treatment target of 5 ng/L. These estimates were used to inform ongoing operational and maintenance (O&M) costs associated with treatment alternatives implementing media treatment, including Alternatives 1a, 1b, 2a, 5a, 6a, and 6b. These alternatives assume the media vessels are operated in a lead-lag configuration. The modeling and other approaches described below were used to estimate breakthrough across the lead vessel only, assuming that the lead granular activated carbon (GAC), anion exchange (AIX) resin, or modified clay media bed would be exchanged after breakthrough. In effect, this assumes that the lag media bed would have minimal PFAS loading allowing for a more predictable changeout frequency. This is likely a conservative approach and that in actual practice the lead media bed may be operated past the first detection of PFAS breakthrough.

In the following sections, the term “bed volumes” will be used to describe the volumetric throughput of water treated through a fixed bed media vessel until a specific treatment criterion is reached. Bed volumes are a unitless measure reflecting the volume of water treated and is calculated as the total volume of water treated divided by the empty bed volume of the fixed media bed.

There are uncertainties in the estimated media replacement frequencies used in this study, and sites seeking to implement media sorption for PFAS removal should conduct pilot studies to accurately predict media replacement frequencies based on site-specific water quality and PFAS removal goals.

2 Background on Predicting PFAS Breakthrough of Fixed Bed Vessels

Sorption media in fixed bed columns requires replacement, regeneration, or reactivation once its sorption capacity is used up and breakthrough of target PFAS in vessel effluent exceed acceptable concentrations. Breakthrough times vary with the chemistry of specific PFAS targeted and with the chemical properties of the water. In general, PFAS with long, hydrophobic carbon chains (such as perfluoroocatanoic acid, PFOA) are removed more effectively than short chains (such as perfluorobutanoic acid, PFBA). Sorption media also tend to have a greater affinity for PFAS with sulfonic acid polar groups than PFAS with carboxylic acid polar groups. Natural organic matter (NOM) competes with PFAS for sorption sites and thus adversely impacts performance and increases the media usage rate. The concentration of major anions in the water will also influence the performance of AIX due to a similar competition mechanism. Breakthrough defines media usage rates and, hence, influences operating costs for the removal process.

For this study, PFAS removal through GAC was predicted by using an advection-dispersion-sink mass balance model (commonly applied for sorption processes in fixed bed reactors) coupled with a homogenous surface diffusion model (HSDM) as a way to estimate the frequency of GAC change-out (Burkhardt et al., 2022). Implementation of this model is detailed in Section 3.

In contrast to GAC, AIX resins and modified clay media have limited studies in the literature that have applied mechanistic-based models for PFAS removal. Dixit et al. provided batch isotherm data needed to solve a coupled mass balance and HSDM for PFAS removal through AIX resin columns (Dixit et al., 2021); however, the resin used by Dixit et al. has shown relatively low capacity for PFAS removal than other PFAS-specific AIX resins in flow-through column experiments (Ellis et al., 2022). Studies using other AIX resins with sufficient isotherm data to enable mechanistic modeling of PFAS removal through PFAS-specific AIX resins were not identified. There are examples of studies applying the Thomas model to describe PFAS removal through AIX resins (Ellis et al., 2022; Schaefer, Nguyen, et al., 2019). The Thomas model, however, is a descriptive model rather than a mechanistic model, so the fit parameters are specific to the study conditions (such as influent concentration, flow rate, and bed length) with limited applicability across projects and sites (Myers et al., 2023). Thus, modeling to predict PFAS removal and breakthrough through AIX resins was not completed. Instead, the bed volumes to breakthrough were estimated based on literature column studies (Boyer et al., 2021; Ellis et al., 2022; Franke et al., 2019; Woodard et al., 2017) and vendor input. For Alternatives 6a and 6b, in which AIX resin beds follow lead-lag GAC beds, modeling was not completed due to overall complexity in modeling AIX breakthrough with variable influent concentrations resulting from upstream GAC performance and breakthrough. Rather, for these two alternatives, it was assumed that the AIX resin beds are operated for one year between media change-out events.

For modified clay media, there are examples of flow-through column experiments in peer-reviewed literature (Grieco et al., 2021; Medina et al., 2022; Najm et al., 2021) as well as batch kinetic and isotherm tests (Jiang et al., 2022; Yan et al., 2020); however, no publicly available datasets were found showing successful development of a mechanistic breakthrough model for PFAS removal in a fixed bed system with modified clay media. Instead, bed volumes to breakthrough estimates were informed by literature column studies (Grieco et al., 2021; Medina et al., 2022; Najm et al., 2021) and vendor input, similar to the approach taken for AIX resin beds.

The approach taken to estimate the bed volumes to breakthrough for each alternative is summarized below in Table 1.

Table 1 Approach taken to estimate the change-out frequency of sorption media

Alternative	Waste Streams	Media	Approach to estimate media change-out frequency
1a	WRRF Effluent, Landfill Leachate, or Compost Contact Water	GAC	HSDM Modeling
1b	RO Concentrate of Landfill Leachate	GAC	HSDM Modeling
2a	WRRF Effluent	AIX	Publicly available breakthrough curves and vendor input
5a	Landfill Leachate or Compost Contact Water	Modified Clay	Publicly available breakthrough curves and vendor input
6a	WRRF Effluent	GAC	HSDM Modeling
		AIX	Assumed one year of operation
6b	RO Concentrate of WRRF Effluent	GAC	HSDM Modeling
		AIX	Assumed one year of operation

3 GAC Breakthrough Modeling

3.1 Model Background

For PFAS adsorption onto GAC, breakthrough was characterized using the mechanistic HSDM with PFAS partitioning between the media and water described by the Freundlich isotherm,

$$q = KC^{1/n} \tag{1}$$

in which q is the media-phase (i.e., surface) concentration and C is the water-phase concentration at equilibrium. The Freundlich isotherm parameters are determined either in suspended batch tests or by curve-fitting the breakthrough curve for a column study.

Both breakthrough modeling approaches allow calculation of breakthrough curves for individual PFAS for varying influent concentrations and reactor configurations. However, the application of either approach has two implicit assumptions. First, the modeled PFAS concentrations are within the range of concentrations for which the Freundlich isotherm were developed. Second, the chemical parameters of the water (e.g., NOM concentration and major anion concentrations) are similar to the test conditions associated with development of the Freundlich or Langmuir isotherm.

The water-phase concentration of an individual PFAS with distance in the column can be characterized by the following equation (Crittenden et al., 1986):

$$\frac{\partial C}{\partial t} = D_x \frac{\partial^2 C}{\partial x^2} - u_x \frac{\partial C}{\partial x} - \frac{3K_f(1-\varepsilon)}{\varepsilon R}\left[C - C_s\right] \tag{3}$$

in which C is the water-phase concentration, t is time, D_x is the dispersion coefficient, x is distance, u_x is the linear water velocity, K_f is the water-film mass transfer coefficient, ε is the bed porosity, R is the radius of a media particle, and C_s is the water-phase concentration at the surface of the media in equilibrium with the media-phase concentration. The initial condition for equation (3) is $C = 0$ at all points within the bed. The Danckwerts boundary conditions apply at the column's inlet and outlet:

$$u_x\left(C_{inf} - C\right)\Big|_{x=0} = -D_x \frac{\partial C}{\partial x}\bigg|_{x=0} \tag{4}$$

$$\frac{\partial C}{\partial x}\bigg|_{x=L} = 0 \tag{5}$$

in which C_{inf} is the influent water-phase PFAS concentration and L is the length (or depth) of the packed-bed.

The media-phase concentration of an individual PFAS with radial distance from the center of the spherical media can be described by the HSDM with Freundlich partitioning:

$$\frac{\partial q}{\partial t} = \frac{1}{r^2}\frac{\partial}{\partial r}\left(r^2 D_s \frac{\partial q}{\partial r}\right) \tag{6}$$

in which D_s is the surface diffusion coefficient. The initial condition for equation (6) is $q = 0$ at all points within the media particle. Equation 6 has two boundary conditions. First, at the center of the spherical media particle, the radial concentration gradient is 0 (Equation 7).

$$\frac{\partial q}{\partial r}\bigg|_{r=0} = 0 \tag{7}$$

Second, the mass flux from the bulk water phase to the surface of the media equals the flux into the media (Equation 8):

$$K_f\left[C - C_s\right] = \rho_m D_s \frac{\partial q}{\partial r}\bigg|_{r=R} \tag{8}$$

in which ρ_m is the density of a media particle. From equation 1, the water-phase concentration at the surface of the media is:

$$C_s = \left(\frac{q}{K}\right)^n\bigg|_{r=R} . \tag{9}$$

3.2 Model Input Parameters

3.2.1 System Flow and GAC Vessel Sizing

The modeled flow and GAC vessel size were selected to achieve an empty bed contact time (EBCT) of 15 minutes per vessel for each alternative.

3.2.2 Mass Transfer Parameters

For this work, the calculation of water-film mass transfer coefficient K_f followed Jarvie et al. (2005). The water-phase diffusion coefficient D_w for the individual PFAS were obtained from Schaefer et al. (2019). The surface diffusivities of PFAS on GAC (D_s) were calculated using the approach presented in Jarvie et al. (2005).

3.2.3 Freundlich Parameters

The recent literature was reviewed for Freundlich isotherms describing the accumulation of individual PFAS onto GAC. The available Freundlich isotherms were screened based on the following criteria:

- the isotherms should reflect individual PFAS concentrations in the ng/L to µg/L range, and not the mg/L range;
- the isotherms for individual PFAS should be developed under exposure to a mixture of PFAS, so that some degree of competitive effects is intrinsically included;
- the mixture of PFAS should include the short chain PFAS such as PFBA and PFBS; and
- the isotherms should be developed with exposure to natural organic matter (NOM), so that some degree of competitive effects is intrinsically included.

The GAC Freundlich isotherm parameters were obtained from a pilot-scale column study treating the effluent from a biological reactor at a water utility (Burkhardt et al., 2022). The selected data set used Calgon F400 as the GAC. The GAC columns were exposed to a total organic carbon (TOC) concentration of about 2.2 mg/L. The selected GAC Freundlich isotherm parameters are provided in Table 2.

Table 2 GAC Freundlich Isotherm Parameters (Calgon F400) (Burkhardt et al., 2022)

PFAS	K [(µg/g)(L/µg)$^{1/n}$]	1/n [unitless]
PFBA	10.6	0.70
PFBS	9.17	0.71
PFHxA	56.0	1.0
PFHxS	21.6	0.85
PFOA	62.3	1.0
PFOS	120	1.0

3.2.4 Influent PFAS Concentrations

PFAS concentrations used in the coupled mass balance and HSDM are summarized in Table 3. For all PFAS and modeling scenarios, the targeted breakthrough concentration was 5 ng/L.

For alternatives implementing RO (Alternatives 1b and 6b), an 85% water recovery was assumed for wastewater effluent and 65% recovery for landfill leachate. The PFAS rejection efficiency was assumed to be 100%.

Table 3 Assumed PFAS Concentrations for Each Waste Stream Used in Breakthrough Modeling

PFAS	Wastewater Effluent	Wastewater Effluent, Concentrate from RO	Landfill Leachate	Landfill Leachate, Concentrate from RO	Compost Contact Water
	Alts 1a, 6a	Alts 6b	Alt 1a	Alt 1b	Alt 1a
PFBA	15	75	950	2,714	450
PFBS	15	75	250	714	10
PFHxA	30	150	1,500	4,286	500
PFHxS	5	25	350	1,000	10
PFOA	40	200	900	2,571	30
PFOS	5	25	150	429	20
6:2 FTS	5	25	150	429	non-detect
N-EtFOSAA	non-detect	non-detect	150	429	non-detect

All concentrations in nanograms per liter (ng/L).

3.3 Model Implementation

Breakthrough curves were obtained from a split-operator finite-difference algorithm in which reactor transport (Equation 3) was solved separately from diffusive transport within the spherical GAC particle (Equation 6). The split-operator approach is well suited for situations with sharp breakthrough curves (Miller & Rabideau, 1993). The algorithm's accuracy was verified by comparing to the analytical solution for linear adsorption isotherms (Rosen, 1954).

3.4 GAC Breakthrough Modeling Results

The HSDM model results are summarized in Table 4.

Table 4 Estimated Bed Volumes to Breakthrough Across GAC Media Beds

PFAS	Wastewater Effluent	Wastewater Effluent, Concentrate from RO	Landfill Leachate	Landfill Leachate, Concentrate from RO	Compost Contact Water
	Alts 1a, 6a	Alt 6b	1a	1b	1a
PFBA	19,500	10,400	**5,100**	**3,600**	**6,400**
PFBS	**15,700**	**8,100**	6,000	4,300	18,600
PFHxA	21,700	15,000	10,800	9,100	12,800
PFHxS	N/A[2]	14,000	6,900	5,200	22,500
PFOA	20,500	12,900	10,000	7,900	22,200
PFOS	N/A[2]	42,500	29,600	23,700	49,100
6:2 FTS	N/A[1]	N/A[1]	N/A[1]	N/A[1]	N/A[1]
N-EtFOSAA	N/A[1]	N/A[1]	N/A[1]	N/A[1]	N/A[1]

All values are unitless bed volumes to breakthrough across a packed bed of Calgon F400 GAC.
The bed volumes to breakthrough used in O&M estimates are **bolded**.
N/A=Not available

[1] Freundlich isotherm parameters were not reported for 6:2 FTS or N-EtFOSAA in Burkhardt et al. (2022).

[2] Model runs were not long enough to observe breakthrough target for PFHxS and PFOS. These two PFAS are not expected to limit the GAC change-out frequency.

4 Bed Volume to Breakthrough Estimates

The bed volumes to breakthrough used to inform operational cost estimates are summarized in Table 5.

Table 5 Summary of estimated bed volumes to breakthrough used to inform operational cost estimates

Alternative	Waste Streams	Media	Bed Volumes to Breakthrough	Basis
1a	WRRF Effluent	GAC	10,000	HSDM Modeling[1]
	Landfill Leachate	GAC	5,100	HSDM Modeling
	Compost Contact Water	GAC	6,400	HSDM Modeling
1b	RO Concentrate of Landfill Leachate	GAC	3,600	HSDM Modeling
2a	WRRF Effluent	AIX Resin	20,000	Ellis et al. (2022)
5a	Landfill Leachate	Modified Clay	5,000	Vendor input and limited literature
	Compost Contact Water	Modified Clay	10,000	Vendor input and limited literature
6a	WRRF Effluent	GAC	10,000	HSDM Modeling[1]
		AIX Resin	140,000	Assumed one year of operation (140,000 bed volumes = 1 year)
6b	RO Concentrate of WRRF Effluent	GAC	8,100	HSDM Modeling
		AIX Resin	140,000	Assumed one year of operation (140,000 bed volumes = 1 year)

[1] HSDM modeling estimated bed volumes to breakthrough to be 15,700. Based on referenced literature (see Appendix C), the maximum bed volumes to breakthrough for GAC was assumed to be 10,000 bed volumes.

5 References

Boyer, T. H., Fang, Y., Ellis, A., Dietz, R., Choi, Y. J., Schaefer, C. E., Higgins, C. P., & Strathmann, T. J. (2021). Anion exchange resin removal of per- and polyfluoroalkyl substances (PFAS) from impacted water: A critical review. *Water Research*, *200*, 117244. https://doi.org/10.1016/j.watres.2021.117244

Burkhardt, J. B., Burns, N., Mobley, D., Pressman, J. G., Magnuson, M. L., & Speth, T. F. (2022). Modeling PFAS Removal Using Granular Activated Carbon for Full-Scale System Design. *Journal of Environmental Engineering*, *148*(3), 04021086. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001964

Crittenden, J. C., Hutzler, N. J., Geyer, D. G., Oravitz, J. L., & Friedman, G. (1986). Transport of Organic Compounds With Saturated Groundwater Flow: Model Development and Parameter Sensitivity. *Water Resources Research*, *22*(3), 271–284. https://doi.org/10.1029/WR022i003p00271

Dixit, F., Barbeau, B., Lompe, K. M., Kheyrandish, A., & Mohseni, M. (2021). Performance of the HSDM to predict competitive uptake of PFAS, NOM and inorganic anions by suspended ion exchange processes. *Environ. Sci.: Water Res. Technol.*, *7*(8), 1417–1429. https://doi.org/10.1039/D1EW00145K

Ellis, A. C., Liu, C. J., Fang, Y., Boyer, T. H., Schaefer, C. E., Higgins, C. P., & Strathmann, T. J. (2022). Pilot study comparison of regenerable and emerging single-use anion exchange resins for treatment of groundwater contaminated by per- and polyfluoroalkyl substances (PFASs). *Water Research*, *223*, 119019. https://doi.org/10.1016/j.watres.2022.119019

Franke, V., Schäfers, M. D., Joos Lindberg, J., & Ahrens, L. (2019). Removal of per- and polyfluoroalkyl substances (PFASs) from tap water using heterogeneously catalyzed ozonation. *Environ. Sci.: Water Res. Technol.*, *5*(11), 1887–1896. https://doi.org/10.1039/C9EW00339H

Grieco, S. A., Chang, J., Maio, E. (Lily) Y., & Hwang, M. (2021). Comparing conventional and emerging adsorbents for per- and polyfluoroalkyl substances: Kinetic, equilibrium, and column experiments. *AWWA Water Science*, *3*(6). https://doi.org/10.1002/aws2.1256

Jarvie, M. E., Hand, D. W., Bhuvendralingam, S., Crittenden, J. C., & Hokanson, D. R. (2005). Simulating the performance of fixed-bed granular activated carbon adsorbers: Removal of synthetic organic chemicals in the presence of background organic matter. *Water Research*, *39*(11), 2407–2421. https://doi.org/10.1016/j.watres.2005.04.023

Jiang, T., Zhang, W., Ilango, A. K., Feldblyum, J. I., Wei, Z., Efstathiadis, H., Yigit, M. V., & Liang, Y. (2022). Surfactant-Modified Clay for Adsorption of Mixtures of Per- and Polyfluoroalkyl Substances (PFAS) in Aqueous Solutions. *ACS Applied Engineering Materials*, acsaenm.2c00096. https://doi.org/10.1021/acsaenm.2c00096

Medina, R., Pannu, M. W., Grieco, S. A., Hwang, M., Pham, C., & Plumlee, M. H. (2022). Pilot-scale comparison of granular activated carbons, ion exchange, and alternative adsorbents for per- and polyfluoroalkyl substances removal. *AWWA Water Science*, *4*(5), e1308. https://doi.org/10.1002/aws2.1308

Miller, C. T., & Rabideau, A. J. (1993). Development of split-operator, Petrov-Galerkin Methods to simulate transport and diffusion problems. *Water Resources Research*, *29*(7), 2227–2240. https://doi.org/10.1029/93WR00528

Myers, T. G., Cabrera-Codony, A., & Valverde, A. (2023). On the development of a consistent mathematical model for adsorption in a packed column (and why standard models fail). *International Journal of Heat and Mass Transfer*, *202*, 123660. https://doi.org/10.1016/j.ijheatmasstransfer.2022.123660

Najm, I., Gallagher, B., Vishwanath, N., Blute, N., Gorzalski, A., Feffer, A., & Richardson, S. (2021). Per- and polyfluoroalkyl substances removal with granular activated carbon and a specialty adsorbent: A case study. *AWWA Water Science*, *3*(5). https://doi.org/10.1002/aws2.1245

Rosen, J. B. (1954). General Numerical Solution for Solid Diffusion in Fixed Beds. *Industrial & Engineering Chemistry*, *46*(8), 1590–1594. https://doi.org/10.1021/ie50536a026

Schaefer, C. E., Drennan, D. M., Tran, D. N., Garcia, R., Christie, E., Higgins, C. P., & Field, J. A. (2019). Measurement of Aqueous Diffusivities for Perfluoroalkyl Acids. *Journal of Environmental Engineering*, *145*(11), 06019006. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001585

Schaefer, C. E., Nguyen, D., Ho, P., Im, J., & LeBlanc, A. (2019). Assessing Rapid Small-Scale Column Tests for Treatment of Perfluoroalkyl Acids by Anion Exchange Resin. *Industrial & Engineering Chemistry Research*, *58*(22), 9701–9706. https://doi.org/10.1021/acs.iecr.9b00858

Woodard, S., Berry, J., & Newman, B. (2017). Ion exchange resin for PFAS removal and pilot test comparison to GAC. *Remediation Journal*, *27*(3), 19–27. https://doi.org/10.1002/rem.21515

Yan, B., Munoz, G., Sauvé, S., & Liu, J. (2020). Molecular mechanisms of per- and polyfluoroalkyl substances on a modified clay: A combined experimental and molecular simulation study. *Water Research*, *184*, 116166. https://doi.org/10.1016/j.watres.2020.116166

Appendix E

Detailed Design Basis and Cost Tables

Appendix E

Alternative 1a (GAC, Incineration) – Municipal Wastewater

Appendix E: Table 1a - WW	**Alternative 1a - GAC, Incineration**			**Units**	**Notes**
Waste stream	Wastewater	Wastewater	Wastewater	--	
Design flow rate	0.1	1	10	MGD	
Design flow rate	70	700	7,000	gpm	Rounded to one significant digit.
Pressure Vessels					
Vessel capacity	6,000	20,000	60,000	lb	
# of lead/lag vessel trains	1	3	9	--	
# of vessels	2	6	18	--	
Media type	12x40	12x40	12x40	--	Reagglomerated, coal-based GAC.
Media apparent density	0.54	0.54	0.54	g/cm3	
Media volume, per vessel	150	470	1,600	ft3	Calculated based on target EBCT. Rounded to two significant digits.
Media weight, per vessel	5,100	16,000	54,000	lb	Calculated based on target EBCT. Rounded to two significant digits.
Empty-bed contact time, per vessel	15	15	15	minutes	
Vessel diameter	6.0	12	14	ft	Showing two significant digits.
Media bed depth	5.0	4.1	10.1	ft	Showing two significant digits.
Hydraulic loading rate	2.5	2.1	5.1	gpm/ft2	Showing two significant digits.
Estimated footprint, first train	800	2,400	3,300	ft2	Rounded to the nearest 100 ft2.
Estimated footprint, additional train	600	1,300	2,100	ft2	Rounded to the nearest 100 ft2.
Driving lane	No	Yes	Yes	--	
Pipe gallery	No	Yes	Yes	--	
Lab and bathroom	No	Yes	Yes	--	
Estimated footprint, total	800	5,000	20,100	ft2	Rounded to the nearest 100 ft2.
Estimated time to breakthrough	10,000	10,000	10,000	bed volumes	Based on modelled breakthrough of PFBS at 5 ng/L.
Estimated changeout frequency	100	100	100	days	Rounded down to the nearest 10 days.
# Trucks per event (virgin GAC)	1	2	13	--	Rounded to nearest integer.
# Trucks per event (spent GAC)	1	3	25	--	Assumes spent media is 2x weight after use, requiring 2x trucks to remove than deliver. Rounded to nearest integer.

Appendix E: Table 1a - WW	Alternative 1a - GAC, Incineration			Units	Notes
Change-out events per year	3.7	3.7	3.7	--	Rounded to two significant digits.
Estimated media usage rate	22,000	220,000	2,000,000	lbs/year	Rounded to two significant digits.
Estimated media usage rate	0.60	0.60	0.55	lbs/1000-gal	Pounds per 1,000 gallons treated.

PREPARED BY: BARR ENGINEERING COMPANY

SHEET:	1	OF	1	
BY:	RRM	DATE:	12/8/2022	
CHECKED BY:	AJM2	DATE:	3/3/2023	
APPROVED BY:	KMB	DATE:	3/20/2023	

ENGINEER'S OPINION OF PROBABLE PROJECT COST

PROJECT: Evaluation of Current Alternatives and Estimated Costs for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water
LOCATION: MINNESOTA
PROJECT #: 23621432
OPINION OF COST - SUMMARY

ISSUED:	DRAFT 1	DATE:	11/11/2022
ISSUED:	DRAFT 2	DATE:	12/22/2022
ISSUED:	FINAL	DATE:	3/24/2023
ISSUED:		DATE:	

Engineer's Opinion of Probable Capital Cost
Municipal Wastewater - Alternative 1a

Total Treatment Flow Rate			Units
Low	Middle	High	
0.1	1	10	MGD
70	700	7000	GPM

			LOW FLOW RATE			MIDDLE FLOW RATE			HIGH FLOW RATE			
Cat. No.	**ITEM DESCRIPTION**	**UNIT**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**NOTES**
A	General Conditions	LS	1	$184,000	$184,000	1	$1,063,000	$1,063,000	1	$6,104,000	$6,104,000	1,2,3,4
B	PFAS Separation Equipment	LS	1	$320,000	$320,000	1	$1,767,000	$1,767,000	1	$12,096,000	$12,096,000	1,2,3,4
C	Piping and Appurtenances	LS	1	$160,000	$160,000	1	$884,000	$884,000	1	$6,048,000	$6,048,000	1,2,3,4
D	Electrical and Instrumentation/Controls	LS	1	$64,000	$64,000	1	$354,000	$354,000	1	$2,420,000	$2,420,000	1,2,3,4
E	Treatment Building	SF	800	$500	$400,000	5,000	$500	$2,500,000	20,100	$500	$10,050,000	1,2,3,4
F	Site Work	LS	1	$40,000	$40,000	1	$250,000	$250,000	1	$1,005,000	$1,005,000	1,2,3,4
G	Installation (Equipment and Piping)	LS	1	$240,000	$240,000	1	$1,326,000	$1,326,000	1	$9,072,000	$9,072,000	1,2,3,4
	CONSTRUCTION SUBTOTAL				$1,410,000			$8,140,000			$46,800,000	1,2,3,4,5
	CONSTRUCTION CONTINGENCY			25%	$350,000		25%	$2,040,000		25%	$11,700,000	1,4,5
	ESTIMATED CONSTRUCTION COST				$1,760,000			$10,180,000			$58,500,000	1,2,3,4,5
	ESTIMATED TOTAL PROJECT COST				**$1,800,000**			**$10,200,000**			**$58,500,000**	1,2,3,4,6
	ESTIMATED ACCURACY RANGE	-30%			$1,300,000			$7,200,000			$41,000,000	4,6
		50%			$2,700,000			$15,300,000			$87,800,000	4,6

Notes

[1] Limited design work completed.

[2] Quantities based on design work completed.

[3] Unit prices based on information available at this time.

[4] This feasibility-level (Class 5, 0-2% design completion per AACE International Recommended Practice No. 17R-97) cost estimate is based on preliminary designs, quantities and unit prices. Costs will change with further design. Time value-of-money escalation costs are not included. Contingency is an allowance for the net sum of costs that will be in the Final Total Project Cost at the time of the completion of design, but are not included at this level of project definition. The estimated accuracy range for the Total Project Cost as the project is defined is -30% to +50%. The accuracy range is based on professional judgement considering the level of design completed, the complexity of the project and the uncertainties in the project as scoped. The contingency and the accuracy range are not intended to include costs for future scope changes that are not part of the project as currently scoped or costs for risk contingency. Operation and maintenance costs are not included.

[5] Estimate costs are reported to nearest ten thousand dollars.

[6] Estimate costs are rounded up to nearest one hundred thousand dollars. Estimate costs are to construct each alternative. The estimated costs do not include planning, engineering and design, permitting, construction administration, maintenance, monitoring or additional tasks following construction.

PREPARED BY: BARR ENGINEERING COMPANY

SHEET:	1	OF	1		
BY:	AJM2	DATE:	12/9/2022		
CHECKED BY:	RRM	DATE:	2/17/2023		
APPROVED BY:	KMB	DATE:	3/20/2023		

ENGINEER'S OPINION OF PROBABLE PROJECT COST

PROJECT: Evaluation of Current Alternatives and Estimated Costs for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water
LOCATION: MINNESOTA
PROJECT #: 23621432
OPINION OF COST - SUMMARY

ISSUED:	DRAFT 1	DATE:	11/11/2022
ISSUED:	DRAFT 2	DATE:	12/22/2022
ISSUED:	FINAL	DATE:	3/24/2023
ISSUED:		DATE:	

Engineer's Opinion of Probable O&M Cost

Municipal Wastewater - Alternative 1a - GAC

Total Treatment Flow Rate			Units
Low	Middle	High	
0.1	1	10	MGD
70	700	7,000	GPM

				LOW FLOWRATE		MIDDLE FLOWRATE		HIGH FLOWRATE		
Item Num.	Description of O&M Category	Unit of Measure	Unit Rate (USD/unit)	Annual Quantity	Annual Total (USD)	Annual Quantity	Annual Total (USD)	Annual Quantity	Annual Total (USD)	Notes
1	**VARIABLE COSTS**									
1.1	**UTILITIES**				**$ 12,000**		**$ 26,000**		**$ 155,000**	1
1.1.1	Electricity - Process equipment	kWh	$ 0.13	11,700	$ 2,000	43,500	$ 6,000	361,700	$ 49,000	1,2
1.1.2	Electricity - Non-process energy load (building power, HVAC, etc.)	kWh	$ 0.13	63,000	$ 9,000	121,800	$ 17,000	710,400	$ 95,000	1,2
1.1.3	Natural gas - Non-process energy load (heating)	MMBTU	$ 15	30	$ 1,000	180	$ 3,000	700	$ 11,000	1
1.2	**MEDIA REPLACEMENT AND DISPOSAL**				**$ 129,000**		**$ 991,000**		**$ 4,971,000**	1
1.2.1	Media Purchase	LB	$ 2.50	22,200	$ 56,000	222,000	$ 555,000	0	$ -	1
1.2.2	Transportation - New Media Purchase	MILE	$ 3.00	3,300	$ 10,000	6,700	$ 21,000	0	$ -	1
1.2.3	Media disposal - Incineration	LB	$ 0.65	44,400	$ 29,000	444,000	$ 289,000	0	$ -	1
1.2.4	Transportation - Media incineration	MILE	$ 3.00	2,800	$ 9,000	8,400	$ 26,000	0	$ -	1
1.2.5	Media disposal - Reactivation (Destruction + Media)	LB	$ 2.00	0	$ -	0	$ -	1,998,000	$ 3,996,000	1
1.2.6	Transportation - Media reactivation	MILE	$ 3.00	0	$ -	0	$ -	166,500	$ 500,000	1
1.2.7	General Conditions - Service Provider - Mobilization	TRUCK PER EVENT	$ 5,000	5	$ 25,000	20	$ 100,000	95	$ 475,000	1
1.3	**MAINTENANCE**				**$ 19,000**		**$ 104,000**		**$ 668,000**	1
1.3.1	Process equipment maintenance	% equip. purchase	3%	1	$ 17,000	1	$ 91,000	1	$ 617,000	1
1.3.2	General building maintenance	sq foot	$ 2.50	800	$ 2,000	5,000	$ 13,000	20,100	$ 51,000	1
1.4	**ANALYTICAL MONITORING**				**$ 16,000**		**$ 36,000**		**$ 92,000**	1
1.4.1	Monitoring - Monthly samples	SAMPLE+SHIPPING	$ 400	40	$ 16,000	90	$ 36,000	230	$ 92,000	1
Item Num.	Description of O&M Category	Unit of Measure	Annual Salary (USD)	Annual Quantity	Annual Total (USD)	Annual Quantity	Annual Total (USD)	Annual Quantity	Annual Total (USD)	Notes
2	**FIXED COSTS**									
2.1	**O&M LABOR**									
2.1.1	Water Treatment Operator	FTE	$ 100,000	1	$ 100,000	1	$ 100,000	2	$ 200,000	1
2.1.2	Shift Maintenance (Mechanical, Electrical)	FTE	$ 100,000	0.5	$ 50,000	0.5	$ 50,000	1	$ 100,000	1
	Total Workforce	**Base Payroll**			$ 150,000		$ 150,000		$ 300,000	1
		Fringe Rate	0.5		$ 75,000		$ 75,000		$ 150,000	1
		Total Payroll			**$ 225,000**		**$ 225,000**		**$ 450,000**	1
	ESTIMATED TOTAL ANNUAL OPERATING COSTS				**$ 410,000**		**$ 1,400,000**		**$ 6,400,000**	3
	Estimated Uncertainty Range	-30%			$ 290,000		$ 980,000		$ 4,480,000	3,4
		+50%			$ 620,000		$ 2,100,000		$ 9,600,000	3,4
	Unit Cost per 1,000 gallons treated				$ 11.23		$ 3.84		$ 1.75	

Notes

[1] Annual total cost values are rounded up to the nearest thousand USD.

[2] Assumed operating 7 days of operation per week, 365 days per year. Adjusted based on flow. Reference: Electric Power Research Institute and Water Research Foundation. Electricity Use and Management in the Municipal Water Supply and Wastewater Industries. Report 3002001433. November 2013.

[3] Values less than $1 million are rounded to the nearest ten thousand USD, and values greater than or equal to $1 million are rounded up to the nearest one hundred thousand USD.

[4] Operation and maintenance costs are based on a Class 5 capital cost estimate with a +50/-30% uncertainty as applicable for projects at less than 2% of full project definition per AACE International 17R-97. Operation and maintenance costs are also expected to have a +50/-30% uncertainty.

Appendix E

Alternative 1a (GAC, Incineration) – Landfill Leachate / Compost Contact Water

Appendix E: Table 1a - LL/CCW	**Alternative 1a - GAC, Incineration**			**Units**	**Notes**
Waste stream	LL/CCW	LL/CCW	LL/CCW	--	
Design flow rate	0.00144	0.0144	0.14	MGD	
Design flow rate	1	10	100	gpm	Rounded to one significant digit.
Pressure Vessels					
Vessel capacity	90	750	10,000	lb	
# of lead/lag vessel trains	2	1	1	--	
# of vessels	4	2	2	--	
Media type	12x40	12x40	12x40	--	Reagglomerated, coal-based GAC.
Media apparent density	0.54	0.54	0.54	g/cm3	
Media volume, per vessel	2.0	21	210	ft3	Calculated based on target EBCT. Rounded to two significant digits.
Media weight, per vessel	70	710	7,100	lb	Calculated based on target EBCT. Rounded to two significant digits.
Empty-bed contact time, per vessel	15	15	15	minutes	
Vessel diameter	1.2	2.0	8.0	ft	Showing two significant digits.
Media bed depth	0.9	6.4	4.0	ft	Showing two significant digits.
Hydraulic loading rate	0.5	3.2	2.0	gpm/ft2	Showing two significant digits
Estimated footprint, first train	50	140	1,900	ft2	1 and 10 gpm rounded to the nearest 10 ft2. 100 gpm rounded to the nearest 100 ft2.
Estimated footprint, additional train	40	100	1,300	ft2	1 and 10 gpm rounded to the nearest 10 ft2. 100 gpm rounded to the nearest 100 ft2.
Driving lane	No	No	Yes	--	
Pipe gallery	No	No	Yes	--	
Lab and bathroom	No	No	No	--	
Estimated footprint, total	90	140	1,900	ft2	1 and 10 gpm rounded to the nearest 10 ft2. 100 gpm rounded to the nearest 100 ft2.

Appendix E: Table 1a - LL/CCW	Alternative 1a - GAC, Incineration			Units	Notes
Estimated Time to Breakthrough - LL					
Estimated time to breakthrough	5,100	5,100	5,100	bed volumes	Based on modelled breakthrough of PFBS at 5 ng/L.
Estimated changeout frequency	50	50	50	days	Rounded down to the nearest 10 days.
# Trucks per event (virgin GAC)	1	1	1	--	Rounded to nearest integer.
# Trucks per event (spent GAC)	1	1	1	--	Assumes spent media is 2x weight after use, requiring 2x trucks to remove than deliver. Rounded to nearest integer.
Change-out events per year	7.3	7.3	7.3	--	Rounded to two significant digits.
Estimated media usage rate	1,400	5,500	73,000	lbs/year	Rounded to two significant digits.
Estimated media usage rate	2.66	1.05	1.39	lbs/1000-gal	Pounds per 1,000 gallons treated.
Estimated Time to Breakthrough - CCW					
Estimated time to breakthrough	6,400	6,400	6,400	bed volumes	Based on modelled breakthrough of PFBS at 5 ng/L.
Estimated changeout frequency	60	60	60	days	Rounded down to the nearest 10 days.
# Trucks per event (virgin GAC)	1	1	1	--	Rounded to nearest integer.
# Trucks per event (spent GAC)	1	1	1	--	Assumes spent media is 2x weight after use, requiring 2x trucks to remove than deliver. Rounded to nearest integer.
Change-out events per year	6.1	6.1	6.1	--	Rounded to two significant digits.
Estimated media usage rate	1,100	4,600	61,000	lbs/year	Rounded to two significant digits.
Estimated media usage rate	2.09	0.88	1.16	lbs/1000-gal	Pounds per 1,000 gallons treated.

PREPARED BY: BARR ENGINEERING COMPANY

ENGINEER'S OPINION OF PROBABLE PROJECT COST

PROJECT: Evaluation of Current Alternatives and Estimated Costs for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water
LOCATION: MINNESOTA
PROJECT #: 23621432

OPINION OF COST - SUMMARY

SHEET: 1		OF	1
BY:	RRM	DATE:	12/8/2022
CHECKED BY:	AJM2	DATE:	3/3/2023
APPROVED BY:	KMB	DATE:	3/20/2023
ISSUED:	DRAFT 1	DATE:	11/11/2022
ISSUED:	DRAFT 2	DATE:	12/22/2022
ISSUED:	FINAL	DATE:	3/24/2023
ISSUED:		DATE:	

Engineer's Opinion of Probable Capital Cost Landfill Leachate / Compost Contact Water - Alternative 1a

Total Treatment Flow Rate			Units
Low	Middle	High	
0.00144	0.0144	0.144	MGD
1	10	100	GPM

			LOW FLOW RATE			MIDDLE FLOW RATE			HIGH FLOW RATE			
Cat. No.	**ITEM DESCRIPTION**	**UNIT**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**NOTES**
A	General Conditions	LS	1	$16,000	$16,000	1	$42,000	$42,000	1	$294,000	$294,000	1,2,3,4
B	PFAS Separation Equipment	LS	1	$21,000	$21,000	1	$82,000	$82,000	1	$372,000	$372,000	1,2,3,4
C	Piping and Appurtenances	LS	1	$11,000	$11,000	1	$41,000	$41,000	1	$186,000	$186,000	1,2,3,4
D	Electrical and Instrumentation/Controls	LS	1	$5,000	$5,000	1	$17,000	$17,000	1	$75,000	$75,000	1,2,3,4
E	Treatment Building	SF	90	$500	$45,000	140	$500	$70,000	1,900	$500	$950,000	1,2,3,4
F	Site Work	LS	1	$5,000	$5,000	1	$7,000	$7,000	1	$95,000	$95,000	1,2,3,4
G	Installation (Equipment and Piping)	LS	1	$16,000	$16,000	1	$62,000	$62,000	1	$279,000	$279,000	1,2,3,4
	CONSTRUCTION SUBTOTAL				$120,000			$320,000			$2,250,000	1,2,3,4,5
	CONSTRUCTION CONTINGENCY			25%	$30,000		25%	$80,000		25%	$560,000	1,4,5
	ESTIMATED CONSTRUCTION COST				$150,000			$400,000			$2,810,000	1,2,3,4,5
	ESTIMATED TOTAL PROJECT COST				**$200,000**			**$400,000**			**$2,900,000**	1,2,3,4,6
	ESTIMATED ACCURACY RANGE	-30%			$200,000			$300,000			$2,100,000	4,6
		50%			$300,000			$600,000			$4,400,000	4,6

Notes

[1] Limited design work completed.

[2] Quantities based on design work completed.

[3] Unit prices based on information available at this time.

[4] This feasibility-level (Class 5, 0-2% design completion per AACE International Recommended Practice No. 17R-97) cost estimate is based on preliminary designs, quantities and unit prices. Costs will change with further design. Time value-of-money escalation costs are not included. Contingency is an allowance for the net sum of costs that will be in the Final Total Project Cost at the time of the completion of design, but are not included at this level of project definition. The estimated accuracy range for the Total Project Cost as the project is defined is -30% to +50%. The accuracy range is based on professional judgement considering the level of design completed, the complexity of the project and the uncertainties in the project as scoped. The contingency and the accuracy range are not intended to include costs for future scope changes that are not part of the project as currently scoped or costs for risk contingency. Operation and maintenance costs are not included.

[5] Estimate costs are reported to nearest ten thousand dollars.

[6] Estimate costs are rounded up to nearest one hundred thousand dollars. Estimate costs are to construct each alternative. The estimated costs do not include planning, engineering and design, permitting, construction administration, maintenance, monitoring or

PREPARED BY: BARR ENGINEERING COMPANY

ENGINEER'S OPINION OF PROBABLE PROJECT COST

PROJECT: Evaluation of Current Alternatives and Estimated Costs for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water
LOCATION: MINNESOTA
PROJECT #: 23621432
OPINION OF COST - SUMMARY

SHEET:	1	OF	1
BY:	AJM2	DATE:	12/9/2022
CHECKED BY:	RRM	DATE:	2/17/2023
APPROVED BY:	KMB	DATE:	3/20/2023
ISSUED:	DRAFT 1	DATE:	11/11/2022
ISSUED:	DRAFT 2	DATE:	12/22/2022
ISSUED:	FINAL	DATE:	3/24/2023
ISSUED:		DATE:	

Engineer's Opinion of Probable O&M Cost

Landfill Leachate - Alternative 1a - GAC

Total Treatment Flow Rate			Units
Low	Middle	High	
0.00144	0.0144	0.144	MGD
1	10	100	GPM

				LOW FLOWRATE		MIDDLE FLOWRATE		HIGH FLOWRATE		
Item Num.	**Description of O&M Category**	**Unit of Measure**	**Unit Rate (USD/unit)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Notes**
1	**VARIABLE COSTS**									
1.1	**UTILITIES**				**$ 10,000**		**$ 10,000**		**$ 13,000**	1
1.1.1	Electricity - Process equipment	kWh	$ 0.13	8,200	$ 2,000	8,700	$ 2,000	13,200	$ 2,000	1,2
1.1.2	Electricity - Non-process energy load (building power, HVAC, etc.)	kWh	$ 0.13	56,500	$ 8,000	57,400	$ 8,000	65,900	$ 9,000	1,2
1.1.3	Natural gas - Non-process energy load (heating)	MMBTU	$ 15	0	$ -	0	$ -	70	$ 2,000	1
1.2	**MEDIA REPLACEMENT AND DISPOSAL**				**$ 118,000**		**$ 134,000**		**$ 390,000**	1
1.2.1	Media Purchase	LB	$ 2.50	1,400	$ 4,000	5,500	$ 14,000	73,000	$ 183,000	1
1.2.2	Transportation - New Media Purchase	MILE	$ 3.00	6,600	$ 20,000	6,600	$ 20,000	6,600	$ 20,000	1
1.2.3	Media disposal - Incineration	LB	$ 0.65	2,800	$ 2,000	11,000	$ 8,000	146,000	$ 95,000	1
1.2.4	Transportation - Media incineration	MILE	$ 3.00	5,500	$ 17,000	5,500	$ 17,000	5,500	$ 17,000	1
1.2.5	General Conditions - Service Provider - Mobilization	TRUCK PER EVENT	$ 5,000	15	$ 75,000	15	$ 75,000	15	$ 75,000	1
1.3	**MAINTENANCE**				**$ 3,000**		**$ 6,000**		**$ 24,000**	1
1.3.1	Process equipment maintenance	% equipment	3%	1	$ 2,000	1	$ 5,000	1	$ 19,000	1
1.3.2	General building maintenance	sq foot	$ 2.50	90	$ 1,000	140	$ 1,000	1,900	$ 5,000	1
1.4	**ANALYTICAL MONITORING**				**$ 24,000**		**$ 16,000**		**$ 16,000**	1
1.4.1	Monitoring - Monthly samples	SAMPLE+SHIPPING	$ 400	60	$ 24,000	40	$ 16,000	40	$ 16,000	1
Item Num.	**Description of O&M Category**	**Unit of Measure**	**Annual Salary (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Notes**
2	**FIXED COSTS**									
2.1	**O&M LABOR**									
2.1.1	Water Treatment Operator	FTE	$ 100,000	0.5	$ 50,000	0.5	$ 50,000	1	$ 100,000	1
2.1.2	Shift Maintenance (Mechanical, Electrical)	FTE	$ 100,000	0.5	$ 50,000	0.5	$ 50,000	0.5	$ 50,000	1
	Total Workforce	**Base Payroll**			$ 100,000		$ 100,000		$ 150,000	1
		Fringe Rate	0.5		$ 50,000		$ 50,000		$ 75,000	1
		Total Payroll			**$ 150,000**		**$ 150,000**		**$ 225,000**	1
	ESTIMATED TOTAL ANNUAL OPERATING COSTS				**$ 310,000**		**$ 320,000**		**$ 700,000**	3
	Estimated Uncertainty Range	-30%			$ 220,000		$ 230,000		$ 490,000	3,4
		+50%			$ 470,000		$ 480,000		$ 1,050,000	3,4
	Unit Cost per 1,000 gallons treated				$ 590		$ 61		$ 13.32	

Notes

[1] Annual total cost values are rounded up to the nearest thousand USD.

[2] Assumed operating 7 days of operation per week, 365 days per year. Adjusted based on flow. Reference: Electric Power Research Institute and Water Research Foundation. Electricity Use and Management in the Municipal Water Supply and Wastewater Industries. Report 3002001433. November 2013.

[3] Values less than $1 million are rounded to the nearest ten thousand USD, and values greater than or equal to $1 million are rounded up to the nearest one hundred thousand USD.

[4] Operation and maintenance costs are based on a Class 5 capital cost estimate with a +50/-30% uncertainty as applicable for projects at less than 2% of full project definition per AACE International 17R-97. Operation and maintenance costs are also expected to have a +50/-30% uncertainty.

PREPARED BY: BARR ENGINEERING COMPANY

SHEET:	1	OF	1	
BY:	AJM2	DATE:	12/9/2022	
CHECKED BY:	RRM	DATE:	2/17/2023	
APPROVED BY:	KMB	DATE:	3/20/2023	

ENGINEER'S OPINION OF PROBABLE PROJECT COST

PROJECT: Evaluation of Current Alternatives and Estimated Costs for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water
LOCATION: MINNESOTA
PROJECT #: 23621432
OPINION OF COST - SUMMARY

ISSUED:	DRAFT 1	DATE:	11/11/2022
ISSUED:	DRAFT 2	DATE:	12/22/2022
ISSUED:	FINAL	DATE:	3/24/2023
ISSUED:		DATE:	

Engineer's Opinion of Probable O&M Cost

Compost Contact Water - Alternative 1a - GAC

Total Treatment Flow Rate			Units
Low	Middle	High	
0.00144	0.0144	0.144	MGD
1	10	100	GPM

				LOW FLOWRATE		MIDDLE FLOWRATE		HIGH FLOWRATE		
Item Num.	**Description of O&M Category**	**Unit of Measure**	**Unit Rate (USD/unit)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Notes**
1	**VARIABLE COSTS**									
1.1	**UTILITIES**				**$ 10,000**		**$ 10,000**		**$ 13,000**	1
1.1.1	Electricity - Process equipment	kWh	$ 0.13	8,200	$ 2,000	8,700	$ 2,000	13,200	$ 2,000	1,2
1.1.2	Electricity - Non-process energy load (building power, HVAC, etc.)	kWh	$ 0.13	56,500	$ 8,000	57,400	$ 8,000	65,900	$ 9,000	1,2
1.1.3	Natural gas - Non-process energy load (heating)	MMBTU	$ 15	0	$ -	0	$ -	70	$ 2,000	1
1.2	**MEDIA REPLACEMENT AND DISPOSAL**				**$ 86,000**		**$ 99,000**		**$ 314,000**	1
1.2.1	Media Purchase	LB	$ 2.50	1,100	$ 3,000	4,600	$ 12,000	61,000	$ 153,000	1
1.2.2	Transportation - New Media Purchase	MILE	$ 3.00	5,500	$ 17,000	5,500	$ 17,000	5,500	$ 17,000	1
1.2.3	Media disposal - Incineration	LB	$ 0.65	2,200	$ 2,000	9,200	$ 6,000	122,000	$ 80,000	1
1.2.4	Transportation - Media incineration	MILE	$ 3.00	4,600	$ 14,000	4,600	$ 14,000	4,600	$ 14,000	1
1.2.5	General Conditions - Service Provider - Mobilization	TRUCK PER EVENT	$ 5,000	10	$ 50,000	10	$ 50,000	10	$ 50,000	1
1.3	**MAINTENANCE**				**$ 3,000**		**$ 6,000**		**$ 24,000**	1
1.3.1	Process equipment maintenance	% equipment	3%	1	$ 2,000	1	$ 5,000	1	$ 19,000	1
1.3.2	General building maintenance	sq foot	$ 2.50	90	$ 1,000	140	$ 1,000	1,900	$ 5,000	1
1.4	**ANALYTICAL MONITORING**				**$ 24,000**		**$ 16,000**		**$ 16,000**	1
1.4.1	Monitoring - Monthly samples	SAMPLE+SHIPPING	$ 400	60	$ 24,000	40	$ 16,000	40	$ 16,000	1
Item Num.	**Description of O&M Category**	**Unit of Measure**	**Annual Salary (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Notes**
2	**FIXED COSTS**									
2.1	**O&M LABOR**									
2.1.1	Water Treatment Operator	FTE	$ 100,000	0.5	$ 50,000	0.5	$ 50,000	1	$ 100,000	1
2.1.2	Shift Maintenance (Mechanical, Electrical)	FTE	$ 100,000	0.5	$ 50,000	0.5	$ 50,000	0.5	$ 50,000	1
	Total Workforce	**Base Payroll**			$ 100,000		$ 100,000		$ 150,000	1
		Fringe Rate	0.5		$ 50,000		$ 50,000		$ 75,000	1
		Total Payroll			**$ 150,000**		**$ 150,000**		**$ 225,000**	1
	ESTIMATED TOTAL ANNUAL OPERATING COSTS				**$ 280,000**		**$ 290,000**		**$ 600,000**	3
	Estimated Uncertainty Range	-30%			$ 200,000		$ 210,000		$ 420,000	3,4
		+50%			$ 420,000		$ 440,000		$ 900,000	3,4
	Unit Cost per 1,000 gallons treated				$ 533		$ 55		$ 11.42	

Notes

[1] Annual total cost values are rounded up to the nearest thousand USD.

[2] Assumed operating 7 days of operation per week, 365 days per year. Adjusted based on flow. Reference: Electric Power Research Institute and Water Research Foundation. Electricity Use and Management in the Municipal Water Supply and Wastewater Industries. Report 3002001433. November 2013.

[3] Values less than $1 million are rounded to the nearest ten thousand USD, and values greater than or equal to $1 million are rounded up to the nearest one hundred thousand USD.

[4] Operation and maintenance costs are based on a Class 5 capital cost estimate with a +50/-30% uncertainty as applicable for projects at less than 2% of full project definition per AACE International 17R-97. Operation and maintenance costs are also expected to have a +50/-30% uncertainty.

Appendix E

Alternative 1b (RO, GAC, Incineration) – Landfill Leachate

Appendix E: Table 1b - LL	**Alternative 1b - RO, Concentrate to GAC and Incineration**			**Units**	**Notes**
Waste stream	LL	LL	LL	--	
Design flow rate	0.0014400	0.0144	0.144	MGD	
Design flow rate	1	10	100	gpm	Rounded to one significant digit.
Membranes					
Recovery	65%	65%	65%	%	Assumed, lower than WW due to higher salinity.
# of skids	1	1	1	--	
# Elements	2	7	54		
Membrane type	RO	RO	RO	--	
Flux	16	16	16	gfd	gfd = gallons per sq. ft. per day.
Concentrate management flow rate	0.0005	0.005	0.05	MGD	
Concentrate management flow rate	1	4	40	gpm	Rounded to one significant digit.
Estimated footprint, total (membranes)	10	300	700	ft2	
Pressure Vessels (GAC)					
Vessel capacity	90	250	2,000	lb	
# of lead/lag vessel trains	2	2	2	--	
# of vessels	4	4	4	--	
Media type	12x40	12x40	12x40	--	Reagglomerated, coal-based GAC.
Media apparent density	0.54	0.54	0.54	g/cm3	
Media volume, per vessel	2	5	41	ft3	Calculated based on target EBCT. Rounded to nearest integer.
Media weight, per vessel	70	170	1,400	lb	Calculated based on target EBCT. Rounded to two significant digits.
Empty-bed contact time, per vessel	15	15	15	minutes	
Vessel diameter	1.2	2.0	5.0	ft	Showing two significant digits.
Media bed depth	0.94	1.3	2.0	ft	Showing two significant digits.
Hydraulic loading rate	0.47	0.64	1.0	gpm/ft2	Showing two significant digits.
Estimated footprint, first train	50	60	900	ft2	1 and 10 gpm rounded to the nearest 10 ft2. 100 gpm rounded to the nearest 100 ft2.

Appendix E: Table 1b - LL	**Alternative 1b - RO, Concentrate to GAC and Incineration**			**Units**	**Notes**
Estimated footprint, additional train	40	50	600	ft2	1 and 10 gpm rounded to the nearest 10 ft2. 100 gpm rounded to the nearest 100 ft2.
Driving lane	No	No	Yes	--	
Pipe gallery	No	No	Yes	--	
Lab and bathroom	No	No	No	--	
Estimated footprint, total (GAC)	90	110	1,500	ft2	1 and 10 gpm rounded to the nearest 10 ft2. 100 gpm rounded to the nearest 100 ft2.
Combined footprint, total	100	410	2,200	ft2	1 and 10 gpm rounded to the nearest 10 ft2. 100 gpm rounded to the nearest 100 ft2.
Estimated time to breakthrough, GAC	3,600	3,600	3,600	bed volumes	Based on modelled breakthrough of PFBS at 5 ng/L.
Estimated changeout frequency, GAC	30	30	30	days	Rounded down to the nearest 10 days.
# Trucks per event (virgin GAC)	1	1	1	--	Rounded to nearest integer.
# Trucks per event (spent GAC)	1	1	1	--	Assumes spent media is 2x weight after use, requiring 2x trucks to remove than deliver. Rounded to nearest integer.
Change-out events per year	13	13	13	--	Rounded to two significant digits.
Estimated media usage rate	2,200	6,100	49,000	lbs/year	Rounded to two significant digits.
Estimated media usage rate	4.19	1.16	0.93	lbs/1000-gal	Pounds per 1,000 gallons treated.

BARR

PREPARED BY: BARR ENGINEERING COMPANY

ENGINEER'S OPINION OF PROBABLE PROJECT COST

PROJECT: Evaluation of Current Alternatives and Estimated Costs for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water
LOCATION: MINNESOTA
PROJECT #: 23621432

OPINION OF COST - SUMMARY

SHEET:	1	OF	1
BY:	RRM	DATE:	12/8/2022
CHECKED BY:	AJM2	DATE:	3/3/2023
APPROVED BY:	KMB	DATE:	3/20/2023
ISSUED:	DRAFT 1	DATE:	11/11/2022
ISSUED:	DRAFT 2	DATE:	12/22/2022
ISSUED:	FINAL	DATE:	3/24/2023
ISSUED:		DATE:	

Engineer's Opinion of Probable Capital Cost

Landfill Leachate - Alternative 1b

Total Treatment Flow Rate			Units
Low	Middle	High	
0.0014	0.014	0.14	MGD
1	10	100	GPM

			LOW FLOW RATE			MIDDLE FLOW RATE			HIGH FLOW RATE			
Cat. No.	**ITEM DESCRIPTION**	**UNIT**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**NOTES**
A	General Conditions	LS	1	$46,000	$46,000	1	$216,000	$216,000	1	$654,000	$654,000	1,2,3,4
B	PFAS Separation Equipment	LS	1	$31,000	$31,000	1	$92,000	$92,000	1	$679,000	$679,000	1,2,3,4
C	Piping and Appurtenances	LS	1	$16,000	$16,000	1	$46,000	$46,000	1	$340,000	$340,000	1,2,3,4
D	Electrical and Instrumentation/Controls	LS	1	$7,000	$7,000	1	$19,000	$19,000	1	$136,000	$136,000	1,2,3,4
E	Treatment Building	SF	410	$500	$205,000	2,200	$500	$1,100,000	4,900	$500	$2,450,000	1,2,3,4
F	Site Work	LS	1	$21,000	$21,000	1	$110,000	$110,000	1	$245,000	$245,000	1,2,3,4
G	Installation (Equipment and Piping)	LS	1	$24,000	$24,000	1	$69,000	$69,000	1	$510,000	$510,000	1,2,3,4
	CONSTRUCTION SUBTOTAL				$350,000			$1,650,000			$5,010,000	1,2,3,4,5
	CONSTRUCTION CONTINGENCY			25%	$90,000		25%	$410,000		25%	$1,250,000	1,4,5
	ESTIMATED CONSTRUCTION COST				$440,000			$2,060,000			$6,260,000	1,2,3,4,5
	ESTIMATED TOTAL PROJECT COST				**$500,000**			**$2,100,000**			**$6,300,000**	1,2,3,4,6
	ESTIMATED ACCURACY RANGE	-30%			$400,000			$1,500,000			$4,500,000	4,6
		50%			$800,000			$3,200,000			$9,500,000	4,6

Notes

[1] Limited design work completed.

[2] Quantities based on design work completed.

[3] Unit prices based on information available at this time.

[4] This feasibility-level (Class 5, 0-2% design completion per AACE International Recommended Practice No. 17R-97) cost estimate is based on preliminary designs, quantities and unit prices. Costs will change with further design. Time value-of-money escalation costs are not included. Contingency is an allowance for the net sum of costs that will be in the Final Total Project Cost at the time of the completion of design, but are not included at this level of project definition. The estimated accuracy range for the Total Project Cost as the project is defined is -50% to +50%. The accuracy range is based on professional judgement considering the level of design completed, the complexity of the project and the uncertainties in the project as scoped. The contingency and the accuracy range are not intended to include costs for future scope changes that are not part of the project as currently scoped or costs for risk contingency. Operation and maintenance costs are not included.

[5] Estimate costs are reported to nearest ten thousand dollars.

[6] Estimate costs are rounded up to nearest one hundred thousand dollars. Estimate costs are to construct each alternative. The estimated costs do not include planning, engineering and design, permitting, construction administration, maintenance, monitoring or additional tasks following construction.

PREPARED BY: BARR ENGINEERING COMPANY

ENGINEER'S OPINION OF PROBABLE PROJECT COST

SHEET:	1	OF	1
BY:	AJM2	DATE:	12/9/2022
CHECKED BY:	RRM	DATE:	2/17/2023
APPROVED BY:	KMB	DATE:	3/20/2023
ISSUED:	DRAFT 1	DATE:	11/11/2022
ISSUED:	DRAFT 2	DATE:	12/22/2022
ISSUED:	FINAL	DATE:	3/24/2023
ISSUED:		DATE:	

PROJECT: Evaluation of Current Alternatives and Estimated Costs for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water

LOCATION: MINNESOTA

PROJECT #: 23621432

OPINION OF COST - SUMMARY

Engineer's Opinion of Probable O&M Cost

Landfill Leachate - Alternative 1b - RO+GAC

Total Treatment Flow Rate			Units
Low	Middle	High	
0.00144	0.0144	0.144	MGD
1	10	100	GPM

				LOW FLOWRATE		MIDDLE FLOWRATE		HIGH FLOWRATE		
Item Num.	**Description of O&M Category**	**Unit of Measure**	**Unit Rate (USD/unit)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Notes**
1	**VARIABLE COSTS**									
1.1	**UTILITIES**				**$ 10,000**		**$ 14,000**		**$ 45,000**	1
1.1.1	Electricity - Process equipment	kWh	$ 0.13	10,600	$ 2,000	32,800	$ 5,000	254,000	$ 34,000	1,2
1.1.2	Electricity - Non-process energy load (building power, HVAC, etc.)	kWh	$ 0.13	56,500	$ 8,000	57,400	$ 8,000	65,900	$ 9,000	1,2
1.1.3	Natural gas - Non-process energy load (heating)	MMBTU	$ 15	0	$ -	10	$ 1,000	80	$ 2,000	1
1.2	**MEDIA REPLACEMENT AND DISPOSAL**				**$ 200,000**		**$ 217,000**		**$ 389,000**	1
1.2.1	Media Purchase	LB	$ 2.50	2,300	$ 6,000	6,500	$ 17,000	52,000	$ 130,000	1
1.2.2	Transportation - New Media Purchase	MILE	$ 3.00	11,700	$ 36,000	11,700	$ 36,000	11,700	$ 36,000	1
1.2.3	Media disposal - Incineration	LB	$ 0.65	4,600	$ 3,000	13,000	$ 9,000	104,000	$ 68,000	1
1.2.4	Transportation - Media incineration	MILE	$ 3.00	9,900	$ 30,000	9,900	$ 30,000	9,900	$ 30,000	1
1.2.5	General Conditions - Service Provider - Mobilization	TRUCK PER EVENT	$ 5,000	25	$ 125,000	25	$ 125,000	25	$ 125,000	1
1.3	**RO CONSUMABLES**				**$ 7,000**		**$ 20,000**		**$ 277,000**	1
1.3.1	Antiscalant	LB	$ 10	10	$ 1,000	100	$ 1,000	1,000	$ 10,000	1
1.3.2	Clean-in-place Chemicals	LB	$ 2	2,400	$ 5,000	8,400	$ 17,000	129,800	$ 260,000	1
1.3.3	RO Element Replacement (5-year Replacement)	Element	$ 700	1	$ 1,000	2	$ 2,000	10	$ 7,000	1
1.4	**MAINTENANCE**				**$ 3,000**		**$ 6,000**		**$ 41,000**	1
1.4.1	Process equipment maintenance	% equipment	3%	1	$ 2,000	1	$ 5,000	1	$ 35,000	1
1.4.2	General building maintenance	sq foot	$ 2.50	100	$ 1,000	400	$ 1,000	2,200	$ 6,000	1
1.5	**ANALYTICAL MONITORING**				**$ 36,000**		**$ 36,000**		**$ 36,000**	1
1.5.1	Monitoring - Monthly samples	SAMPLE+SHIPPING	$ 400	90	$ 36,000	90	$ 36,000	90	$ 36,000	1
Item Num.	**Description of O&M Category**	**Unit of Measure**	**Annual Salary (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Notes**
2	**FIXED COSTS**									
2.1	**O&M LABOR**									
2.1.1	Water Treatment Operator	FTE	$ 100,000	1	$ 100,000	1	$ 100,000	1	$ 100,000	1
2.1.2	Shift Maintenance (Mechanical, Electrical)	FTE	$ 100,000	0.5	$ 50,000	0.5	$ 50,000	0.5	$ 50,000	1
	Total Workforce	**Base Payroll**			$ 150,000		$ 150,000		$ 150,000	1
		Fringe Rate	0.5		$ 75,000		$ 75,000		$ 75,000	1
		Total Payroll			**$ 225,000**		**$ 225,000**		**$ 225,000**	1
	ESTIMATED TOTAL ANNUAL OPERATING COSTS				**$ 490,000**		**$ 520,000**		**$ 1,100,000**	3
	Estimated Uncertainty Range	-30%			$ 350,000		$ 370,000		$ 770,000	3,4
		+50%			$ 800,000		$ 800,000		$ 1,700,000	3,4
	Unit Cost per 1,000 gallons treated				$ 932		$ 99		$ 20.93	

Notes

[1] Annual total cost values are rounded up to the nearest thousand USD.

[2] Assumed operating 7 days of operation per week, 365 days per year. Adjusted based on flow. Reference: Electric Power Research Institute and Water Research Foundation. Electricity Use and Management in the Municipal Water Supply and Wastewater Industries. Report 3002001433. November 2013.

[3] Values less than $1 million are rounded to the nearest ten thousand USD, and values greater than or equal to $1 million are rounded up to the nearest one hundred thousand USD.

[4] Operation and maintenance costs are based on a Class 5 capital cost estimate with a +50/-30% uncertainty as applicable for projects at less than 2% of full project definition per AACE International 17R-97. Operation and maintenance costs are also expected to have a +50/-30% uncertainty.

Appendix E

Alternative 2a (AIX, Incineration) – Municipal Wastewater

Appendix E: Table 2a - WW	Alternative 2a - AIX, Incineration			Units	Notes
Waste stream	Wastewater	Wastewater	Wastewater	--	
Design flow rate	0.1	1	10	MGD	
Design flow rate	70	700	7,000	gpm	Rounded to one significant digit.
Pressure Vessels					
Vessel capacity	38	201	450	ft3	
# of lead/lag vessel trains	2	2	9	--	
# of vessels	4	4	18	--	
Media type	AIX Resin	AIX Resin	AIX Resin	--	Polystyrenic gel.
Media apparent density	42.2	42.2	42.2	lb/ft3	
Media volume, per vessel	19	190	420	ft3	Calculated based on target EBCT. Rounded to two significant digits.
Media weight, per vessel	900	8,100	18,000	lb	Calculated based on target EBCT. Rounded to two significant digits.
Empty-bed contact time, per vessel	4	4	4	minutes	
Vessel diameter	3.0	8.0	12.0	ft	Showing two significant digits.
Media bed depth	2.6	3.7	3.7	ft	Showing two significant digits.
Hydraulic loading rate	5.0	7.0	6.9	gpm/ft2	Showing two significant digits.
Specific flow rate	0.9	1.7	1.7	gpm/ft3	
Estimated footprint, first train	200	2,400	2,900	ft2	Rounded to the nearest 100 ft2.
Estimated footprint, additional train	200	1,200	1,700	ft2	Rounded to the nearest 100 ft2.
Driving lane	No	Yes	Yes	--	
Pipe gallery	No	Yes	Yes	--	
Lab and bathroom	No	Yes	Yes	--	
Estimated footprint, total	400	3,600	16,500	ft2	Rounded to the nearest 100 ft2.
Estimated time to breakthrough	20,000	20,000	20,000	bed volumes	Based on maximum bed volumes treated in Ellis et al. (2022).
Estimated changeout frequency	50	50	50	days	Rounded down to the nearest 10 days.
# Trucks per event (virgin resin)	1	1	5	--	Rounded to nearest integer.

Appendix E: Table 2a - WW	Alternative 2a - AIX, Incineration			Units	Notes
# Trucks per event (spent resin)	1	1	9	--	Assumes spent media is 2x weight after use, requiring 2x trucks to remove than deliver. Rounded to nearest integer.
Change-out events per year	7.3	7.3	7.3	--	Rounded to two significant digits.
Estimated media usage rate	14,000	120,000	1,200,000	lb/year	Rounded to two significant digits.
Estimated media usage rate	0.38	0.33	0.33	lbs/1000-gal	Pounds per 1,000 gallons treated.

PREPARED BY: BARR ENGINEERING COMPANY

SHEET:	1	OF	1	
BY:	RRM	DATE:	12/8/2022	
CHECKED BY:	AJM2	DATE:	3/3/2023	
APPROVED BY:	KMB	DATE:	3/20/2023	

ENGINEER'S OPINION OF PROBABLE PROJECT COST

PROJECT: Evaluation of Current Alternatives and Estimated Costs for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water
LOCATION: MINNESOTA
PROJECT #: 23621432

OPINION OF COST - SUMMARY

ISSUED:	DRAFT 1	DATE:	11/11/2022
ISSUED:	DRAFT 2	DATE:	12/22/2022
ISSUED:	FINAL	DATE:	3/24/2023
ISSUED:		DATE:	

Engineer's Opinion of Probable Capital Cost

Municipal Wastewater - Alternative 2a

Total Treatment Flow Rate			Units
Low	Middle	High	
0.1	1	10	MGD
70	700	7000	GPM

			LOW FLOW RATE			MIDDLE FLOW RATE			HIGH FLOW RATE			
Cat. No.	ITEM DESCRIPTION	UNIT	QUANTITY	UNIT COST	ITEM COST	QUANTITY	UNIT COST	ITEM COST	QUANTITY	UNIT COST	ITEM COST	NOTES
A	General Conditions	LS	1	$78,000	$78,000	1	$995,000	$995,000	1	$6,982,000	$6,982,000	1,2,3,4
B	PFAS Separation Equipment	LS	1	$122,000	$122,000	1	$1,897,000	$1,897,000	1	$15,293,000	$15,293,000	1,2,3,4
C	Piping and Appurtenances	LS	1	$61,000	$61,000	1	$949,000	$949,000	1	$7,647,000	$7,647,000	1,2,3,4
D	Electrical and Instrumentation/Controls	LS	1	$25,000	$25,000	1	$380,000	$380,000	1	$3,059,000	$3,059,000	1,2,3,4
E	Treatment Building	SF	400	$500	$200,000	3,600	$500	$1,800,000	16,500	$500	$8,250,000	1,2,3,4
F	Site Work	LS	1	$20,000	$20,000	1	$180,000	$180,000	1	$825,000	$825,000	1,2,3,4
G	Installation (Equipment and Piping)	LS	1	$92,000	$92,000	1	$1,423,000	$1,423,000	1	$11,470,000	$11,470,000	1,2,3,4
	CONSTRUCTION SUBTOTAL				$600,000			$7,620,000			$53,530,000	1,2,3,4,5
	CONSTRUCTION CONTINGENCY			25%	$150,000		25%	$1,910,000		25%	$13,380,000	1,4,5
	ESTIMATED CONSTRUCTION COST				$750,000			$9,530,000			$66,910,000	1,2,3,4,5
	ESTIMATED TOTAL PROJECT COST				**$800,000**			**$9,600,000**			**$67,000,000**	1,2,3,4,6
	ESTIMATED ACCURACY RANGE	-30%			$600,000			$6,800,000			$46,900,000	4,6
		50%			$1,200,000			$14,400,000			$100,500,000	4,6

Notes

[1] Limited design work completed.

[2] Quantities based on design work completed.

[3] Unit prices based on information available at this time.

[4] This feasibility-level (Class 5, 0-2% design completion per AACE International Recommended Practice No. 17R-97) cost estimate is based on preliminary designs, quantities and unit prices. Costs will change with further design. Time value-of-money escalation costs are not included. Contingency is an allowance for the net sum of costs that will be in the Final Total Project Cost at the time of the completion of design, but are not included at this level of project definition. The estimated accuracy range for the Total Project Cost as the project is defined is -30% to +50%. The accuracy range is based on professional judgement considering the level of design completed, the complexity of the project and the uncertainties in the project as scoped. The contingency and the accuracy range are not intended to include costs for future scope changes that are not part of the project as currently scoped or costs for risk contingency. Operation and maintenance costs are not included.

[5] Estimate costs are reported to nearest ten thousand dollars.

[6] Estimate costs are rounded up to nearest one hundred thousand dollars. Estimate costs are to construct each alternative. The estimated costs do not include planning, engineering and design, permitting, construction administration, maintenance, monitoring or additional tasks following construction.

PREPARED BY: BARR ENGINEERING COMPANY

ENGINEER'S OPINION OF PROBABLE PROJECT COST

PROJECT: Evaluation of Current Alternatives and Estimated Costs for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water
LOCATION: MINNESOTA
PROJECT #: 23621432
OPINION OF COST - SUMMARY

SHEET: 1 OF 1
BY: AJM2 DATE: 12/9/2022
CHECKED BY: RRM DATE: 2/17/2023
APPROVED BY: KMB DATE: 3/20/2023

ISSUED: DRAFT 1 DATE: 11/11/2022
ISSUED: DRAFT 2 DATE: 12/22/2022
ISSUED: FINAL DATE: 3/24/2023
ISSUED: DATE:

Engineer's Opinion of Probable O&M Cost

Municipal Wastewater - Alternative 2a - AIX

Total Treatment Flowrate			Units
Low	**Middle**	**High**	
0.1	1	10	MGD
70	700	7000	GPM

				LOW FLOWRATE		MIDDLE FLOWRATE		HIGH FLOWRATE		
Item Num.	**Description of O&M Category**	**Unit of Measure**	**Unit Rate (USD/unit)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Notes**
1	**VARIABLE COSTS**									
1.1	**UTILITIES**				**$ 12,000**		**$ 25,000**		**$ 153,000**	1
1.1.1	Electricity - Process equipment	kWh	$ 0.13	11,700	$ 2,000	43,500	$ 6,000	361,700	$ 49,000	1,2
1.1.2	Electricity - Non-process energy load (building power, HVAC, etc.)	kWh	$ 0.13	63,000	$ 9,000	121,800	$ 17,000	710,400	$ 95,000	1,2
1.1.3	Natural gas - Non-process energy load (heating)	MMBTU	$ 15	10	$ 1,000	130	$ 2,000	580	$ 9,000	1
1.2	**MEDIA REPLACEMENT AND DISPOSAL**				**$ 218,000**		**$ 1,151,000**		**$ 11,096,000**	1
1.2.1	Media Purchase	CU FT	$ 320	280	$ 90,000	2,770	$ 887,000	27,600	$ 8,832,000	1
1.2.2	Transportation - New Media Purchase	MILE	$ 3.00	6,570	$ 20,000	6,570	$ 20,000	32,900	$ 99,000	1
1.2.3	Media disposal - Incineration	LB	$ 0.65	23,600	$ 16,000	233,800	$ 152,000	2,329,400	$ 1,515,000	1
1.2.4	Transportation - Media incineration	MILE	$ 3.00	5,550	$ 17,000	5,500	$ 17,000	49,900	$ 150,000	1
1.2.5	General Conditions - Service Provider - Mobilization	TRUCK PER EVENT	$ 5,000	15	$ 75,000	15	$ 75,000	100	$ 500,000	1
1.3	**MAINTENANCE**				**$ 8,000**		**$ 106,000**		**$ 822,000**	1
1.3.1	Process equipment maintenance	% equipment	3%	1	$ 7,000	1	$ 97,000	1	$ 780,000	1
1.3.2	General building maintenance	sq foot	$ 2.50	400	$ 1,000	3,600	$ 9,000	16,500	$ 42,000	1
1.4	**ANALYTICAL MONITORING**				**$ 24,000**		**$ 24,000**		**$ 92,000**	1
1.4.1	Monitoring - Monthly samples	SAMPLE+SHIPPING	$ 400	60	$ 24,000	60	$ 24,000	230	$ 92,000	1
Item Num.	**Description of O&M Category**	**Unit of Measure**	**Annual Salary (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Notes**
2	**FIXED COSTS**									
2.1	**O&M LABOR**									
2.1.1	Water Treatment Operator	FTE	$ 100,000	1	$ 100,000	1	$ 100,000	2	$ 200,000	1
2.1.2	Shift Maintenance (Mechanical, Electrical)	FTE	$ 100,000	0.5	$ 50,000	0.5	$ 50,000	1	$ 100,000	1
	Total Workforce	**Base Payroll**			$ 150,000		$ 150,000		$ 300,000	1
		Fringe Rate	0.5		$ 75,000		$ 75,000		$ 150,000	1
		Total Payroll			**$ 225,000**		**$ 225,000**		**$ 450,000**	1
	ESTIMATED TOTAL ANNUAL OPERATING COSTS				**$ 490,000**		**$ 1,600,000**		**$ 12,700,000**	3
	Estimated Uncertainty Range	-30%			$ 350,000		$ 1,120,000		$ 8,890,000	3,4
		+50%			$ 740,000		$ 2,400,000		$ 19,050,000	3,4
	Unit Cost per 1,000 gallons treated				$ 13.42		$ 4.38		$ 3.48	

Notes

[1] Annual total cost values are rounded up to the nearest thousand USD.

[2] Assumed operating 7 days of operation per week, 365 days per year. Adjusted based on flow. Reference: Electric Power Research Institute and Water Research Foundation. Electricity Use and Management in the Municipal Water Supply and Wastewater Industries. Report 3002001433. November 2013.

[3] Values less than $1 million are rounded to the nearest ten thousand USD, and values greater than or equal to $1 million are rounded up to the nearest one hundred thousand USD.

[4] Operation and maintenance costs are based on a Class 5 capital cost estimate with a +50/-30% uncertainty as applicable for projects at less than 2% of full project definition per AACE International 17R-97. Operation and maintenance costs are also expected to have a +50/-30% uncertainty.

Appendix E

Alternative 5a (Modified Clay, Incineration) – Landfill Leachate / Compost Contact Water

Appendix E: Table 5a - LL/CCW	**Alternative 5a - Modified Clay Media, Incineration**			**Units**	**Notes**
Waste stream	LL/CCW	LL/CCW	LL/CCW	--	
Design flow rate	0.00144	0.0144	0.14	MGD	
Design flow rate	1	10	100	gpm	Rounded to one significant digit.
Flowrate per vessel	1	3	25	gpm	
Pressure Vessels					
Vessel capacity	3	5	38	ft3	
# of lead/lag vessel trains	1	3	4	--	
# of vessels	2	6	8	--	
Media type	Modified Clay	Modified Clay	Modified Clay	--	
Media apparent density	46.2	46.2	46.2	lb/ft3	
Media volume, per vessel	2	5	34	ft3	Calculated based on target EBCT. Rounded to nearest integer.
Media weight, per vessel	93	240	1,600	lb	Calculated based on target EBCT. Rounded to two significant digits.
Empty-bed contact time, per vessel	10	10	10	minutes	
Vessel diameter	1.2	1.5	3.0	ft	Showing two significant digits.
Media bed depth	1.3	2.5	4.7	ft	Showing two significant digits.
Hydraulic loading rate	0.9	1.9	3.5	gpm/ft2	Showing two significant digits.
Estimated footprint, first train	50	50	900	ft2	1 and 10 gpm rounded to the nearest 10 ft2. 100 gpm rounded to the nearest 100 ft2.
Estimated footprint, additional train	40	40	400	ft2	1 and 10 gpm rounded to the nearest 10 ft2. 100 gpm rounded to the nearest 100 ft2.
Driving lane	No	No	Yes	--	
Pipe gallery	No	No	Yes	--	
Lab and bathroom	No	No	No	--	

Appendix E: Table 5a - LL/CCW	Alternative 5a - Modified Clay Media, Incineration			Units	Notes
Estimated footprint, total	50	130	2,100	ft2	1 and 10 gpm rounded to the nearest 10 ft2. 100 gpm rounded to the nearest 100 ft2.
Estimated time to breakthrough - LL					
Estimated time to breakthrough	5,000	5,000	5,000	bed volumes	
Estimated changeout frequency	30	30	30	days	Rounded down to the nearest 10 days.
# Trucks per event (virgin modified clay)	1	1	1	--	Rounded to nearest integer.
# Trucks per event (spent modified clay)	1	1	1	--	Assumes spent media is 2x weight after use, requiring 2x trucks to remove than deliver. Rounded to nearest integer.
Change-out events per year	13	13	13	--	Rounded to two significant digits.
Estimated media usage rate	1,300	9,400	84,000	lb/year	Rounded to two significant digits.
Estimated media usage rate	2.47	1.79	1.60	lbs/1000-gal	Pounds per 1,000 gallons treated.
Estimated time to breakthrough - CCW					
Estimated time to breakthrough	10,000	10,000	10,000	bed volumes	
Estimated changeout frequency	60	60	60	days	Rounded down to the nearest 10 days.
# Trucks per event (virgin modified clay)	1	1	1	--	Rounded to nearest integer.
# Trucks per event (spent modified clay)	1	1	1	--	Assumes spent media is 2x weight after use, requiring 2x trucks to remove than deliver. Rounded to nearest integer.
Change-out events per year	6.1	6.1	6.1	--	Rounded to two significant digits.
Estimated media usage rate	600	4,400	40,000	lb/year	Rounded to two significant digits.
Estimated media usage rate	1.14	0.84	0.76	lbs/1000-gal	Pounds per 1,000 gallons treated.

PREPARED BY: BARR ENGINEERING COMPANY

ENGINEER'S OPINION OF PROBABLE PROJECT COST

PROJECT: Evaluation of Current Alternatives and Estimated Costs for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water
LOCATION: MINNESOTA
PROJECT #: 23621432

OPINION OF COST - SUMMARY

SHEET:	1	OF	1
BY:	RRM	DATE:	12/5/2022
CHECKED BY:	AJM2	DATE:	3/3/2023
APPROVED BY:	KMB	DATE:	3/20/2023
ISSUED:	DRAFT 1	DATE:	NA
ISSUED:	DRAFT 2	DATE:	12/22/2022
ISSUED:	FINAL	DATE:	3/24/2023
ISSUED:		DATE:	

Engineer's Opinion of Probable Capital Cost Landfill Leachate / Compost Contact Water - Alternative 5a

Total Treatment Flow Rate			Units
Low	**Middle**	**High**	
0.00144	0.0144	0.144	MGD
1	10	100	GPM

			LOW FLOW RATE			MIDDLE FLOW RATE			HIGH FLOW RATE			
Cat. No.	**ITEM DESCRIPTION**	**UNIT**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**NOTES**
A	General Conditions	LS	1	$13,000	$13,000	1	$28,000	$28,000	1	$262,000	$262,000	1,2,3,4
B	PFAS Separation Equipment	LS	1	$21,000	$21,000	1	$45,000	$45,000	1	$239,000	$239,000	1,2,3,4
C	Piping and Appurtenances	LS	1	$11,000	$11,000	1	$23,000	$23,000	1	$120,000	$120,000	1,2,3,4
D	Electrical and Instrumentation/Controls	LS	1	$5,000	$5,000	1	$9,000	$9,000	1	$48,000	$48,000	1,2,3,4
E	Treatment Building	SF	50	$500	$25,000	130	$500	$65,000	2,100	$500	$1,050,000	1,2,3,4
F	Site Work	LS	1	$3,000	$3,000	1	$7,000	$7,000	1	$105,000	$105,000	1,2,3,4
G	Installation (Equipment and Piping)	LS	1	$16,000	$16,000	1	$34,000	$34,000	1	$180,000	$180,000	1,2,3,4
	CONSTRUCTION SUBTOTAL				$90,000			$210,000			$2,000,000	1,2,3,4,5
	CONSTRUCTION CONTINGENCY			25%	$20,000		25%	$50,000		25%	$500,000	1,4,5
	ESTIMATED CONSTRUCTION COST				$110,000			$260,000			$2,500,000	1,2,3,4,5
	ESTIMATED TOTAL PROJECT COST				**$200,000**			**$300,000**			**$2,500,000**	1,2,3,4,6
	ESTIMATED ACCURACY RANGE	-30%			$200,000			$300,000			$1,800,000	4,6
		50%			$300,000			$500,000			$3,800,000	4,6

Notes

[1] Limited design work completed.

[2] Quantities based on design work completed.

[3] Unit prices based on information available at this time.

[4] This feasibility-level (Class 5, 0-2% design completion per AACE International Recommended Practice No. 17R-97) cost estimate is based on preliminary designs, quantities and unit prices. Costs will change with further design. Time value-of-money escalation costs are not included. Contingency is an allowance for the net sum of costs that will be in the Final Total Project Cost at the time of the completion of design, but are not included at this level of project definition. The estimated accuracy range for the Total Project Cost as the project is defined is -50% to +50%. The accuracy range is based on professional judgement considering the level of design completed, the complexity of the project and the uncertainties in the project as scoped. The contingency and the accuracy range are not intended to include costs for future scope changes that are not part of the project as currently scoped or costs for risk contingency. Operation and maintenance costs are not included.

[5] Estimate costs are reported to nearest ten thousand dollars.

[6] Estimate costs are rounded up to nearest one hundred thousand dollars. Estimate costs are to construct each alternative. The estimated costs do not include planning, engineering and design, permitting, construction administration, maintenance, monitoring or additional tasks following construction.

PREPARED BY: BARR ENGINEERING COMPANY

ENGINEER'S OPINION OF PROBABLE PROJECT COST

SHEET:	1	OF	1
BY:	RRM	DATE:	12/5/2022
CHECKED BY:	RRM	DATE:	2/17/2023
APPROVED BY:	KMB	DATE:	3/20/2023

PROJECT: Evaluation of Current Alternatives and Estimated Costs for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water
LOCATION: MINNESOTA
PROJECT #: 23621432
OPINION OF COST - SUMMARY

ISSUED:	DRAFT 1	DATE:	NA
ISSUED:	DRAFT 2	DATE:	12/22/2022
ISSUED:	FINAL	DATE:	3/24/2023
ISSUED:		DATE:	

Engineer's Opinion of Probable O&M Cost

Landfill Leachate - Alternative 5a - Modified Clay

Total Treatment Flow Rate			Units
Low	Middle	High	
0.00144	0.0144	0.144	MGD
1	10	100	GPM

				LOW FLOWRATE		MIDDLE FLOWRATE		HIGH FLOWRATE		
Item Num.	**Description of O&M Category**	**Unit of Measure**	**Unit Rate (USD/unit)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Notes**
1	**VARIABLE COSTS**									
1.1	**UTILITIES**				**$ 10,000**		**$ 10,000**		**$ 13,000**	1
1.1.1	Electricity - Process equipment	kWh	$ 0.13	8,200	$ 2,000	8,700	$ 2,000	13,200	$ 2,000	1,2
1.1.2	Electricity - Non-process energy load (building power, HVAC, etc.)	kWh	$ 0.13	56,500	$ 8,000	57,400	$ 8,000	65,900	$ 9,000	1,2
1.1.3	Natural gas - Non-process energy load (heating)	MMBTU	$ 15	0	$ -	0	$ -	70	$ 2,000	1
1.2	**MEDIA REPLACEMENT AND DISPOSAL**				**$ 73,000**		**$ 112,000**		**$ 470,000**	1
1.2.1	New Media Purchase	LB	$ 3.50	1,300	$ 5,000	9,400	$ 33,000	84,000	$ 294,000	1
1.2.2	Transportation - New Media Purchase	MILE	$ 3.00	11,700	$ 36,000	11,700	$ 36,000	11,700	$ 36,000	1
1.2.3	Media disposal - Incineration	LB	$ 0.65	2,600	$ 2,000	18,800	$ 13,000	168,000	$ 110,000	1
1.2.4	Transportation - Media incineration	MILE	$ 3.00	9,900	$ 30,000	9,900	$ 30,000	9,900	$ 30,000	1
1.2.5	General Conditions - Service Provider - Mobilization	TRUCK PER EVENT	$ 5,000	25	$ 125,000	25	$ 125,000	25	$ 125,000	1
1.3	**MAINTENANCE & MONITORING**				**$ 3,000**		**$ 4,000**		**$ 19,000**	1
1.3.1	Process equipment maintenance	% equipment	3%	1	$ 2,000	1	$ 3,000	1	$ 13,000	1
1.3.2	General building maintenance	sq foot	$ 2.50	50	$ 1,000	100	$ 1,000	2,100	$ 6,000	1
1.4	**ANALYTICAL MONITORING**				**$ 16,000**		**$ 36,000**		**$ 44,000**	1
1.4.1	Monitoring - Monthly samples	SAMPLE+SHIPPING	$ 400	40	$ 16,000	90	$ 36,000	110	$ 44,000	1
Item Num.	**Description of O&M Category**	**Unit of Measure**	**Annual Salary (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Notes**
2	**FIXED COSTS**									
2.1	**O&M LABOR**									
2.1.1	Water Treatment Operator	FTE	$ 100,000	0.5	$ 50,000	0.5	$ 50,000	1	$ 100,000	1
2.1.2	Shift Maintenance (Mechanical, Electrical)	FTE	$ 100,000	0.5	$ 50,000	0.5	$ 50,000	0.5	$ 50,000	1
	Total Workforce	**Base Payroll**			$ 100,000		$ 100,000		$ 150,000	1
		Fringe Rate	0.5		$ 50,000		$ 50,000		$ 75,000	1
		Total Payroll			**$ 150,000**		**$ 150,000**		**$ 225,000**	1
	ESTIMATED TOTAL ANNUAL OPERATING COSTS				**$ 260,000**		**$ 320,000**		**$ 780,000**	3
	Estimated Uncertainty Range	-30%			$ 190,000		$ 230,000		$ 550,000	3,4
		+50%			$ 390,000		$ 480,000		$ 1,170,000	3,4
	Unit Cost per 1,000 gallons treated				$ 495		$ 61		$ 14.84	

Notes

[1] Annual total cost values are rounded up to the nearest thousand USD.

[2] Assumed operating 7 days of operation per week, 365 days per year. Adjusted based on flow. Reference: Electric Power Research Institute and Water Research Foundation. Electricity Use and Management in the Municipal Water Supply and Wastewater Industries. Report 3002001433. November 2013.

[3] Values less than $1 million are rounded to the nearest ten thousand USD, and values greater than or equal to $1 million are rounded up to the nearest one hundred thousand USD.

[4] Operation and maintenance costs are based on a Class 5 capital cost estimate with a +50/-30% uncertainty as applicable for projects at less than 2% of full project definition per AACE International 17R-97. Operation and maintenance costs are also expected to have a +50/-30% uncertainty.

PREPARED BY: BARR ENGINEERING COMPANY

ENGINEER'S OPINION OF PROBABLE PROJECT COST

SHEET:	1	OF	1	
BY:	RRM	DATE:	12/5/2022	
CHECKED BY:	RRM	DATE:	2/17/2023	
APPROVED BY:	KMB	DATE:	3/20/2023	

PROJECT: Evaluation of Current Alternatives and Estimated Costs for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water
LOCATION: MINNESOTA
PROJECT #: 23621432
OPINION OF COST - SUMMARY

ISSUED:	DRAFT 1	DATE:	NA
ISSUED:	DRAFT 2	DATE:	12/22/2022
ISSUED:	FINAL	DATE:	3/24/2023
ISSUED:		DATE:	

Engineer's Opinion of Probable O&M Cost

Compost Contact Water - Alternative 5a - Modified Clay

Total Treatment Flow Rate			Units
Low	**Middle**	**High**	
0.00144	0.0144	0.144	MGD
1	10	100	GPM

				LOW FLOWRATE		**MIDDLE FLOWRATE**		**HIGH FLOWRATE**		
Item Num.	**Description of O&M Category**	**Unit of Measure**	**Unit Rate (USD/unit)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Notes**
1	**VARIABLE COSTS**									
1.1	**UTILITIES**				**$ 10,000**		**$ 10,000**		**$ 13,000**	1
1.1.1	Electricity - Process equipment	kWh	$ 0.13	8,200	$ 2,000	8,700	$ 2,000	13,200	$ 2,000	1,2
1.1.2	Electricity - Non-process energy load (building power, HVAC, etc.)	kWh	$ 0.13	56,500	$ 8,000	57,400	$ 8,000	65,900	$ 9,000	1,2
1.1.3	Natural gas - Non-process energy load (heating)	MMBTU	$ 15	0	$ -	0	$ -	70	$ 2,000	1
1.2	**MEDIA REPLACEMENT AND DISPOSAL**				**$ 34,000**		**$ 53,000**		**$ 219,000**	1
1.2.1	New Media Purchase	LB	$ 3.50	570	$ 2,000	4,400	$ 16,000	39,000	$ 137,000	1
1.2.2	Transportation - New Media Purchase	MILE	$ 3.00	5,500	$ 17,000	5,500	$ 17,000	5,500	$ 17,000	1
1.2.3	Media disposal - Incineration	LB	$ 0.65	1,100	$ 1,000	8,800	$ 6,000	78,000	$ 51,000	1
1.2.4	Transportation - Media incineration	MILE	$ 3.00	4,600	$ 14,000	4,600	$ 14,000	4,600	$ 14,000	1
1.2.5	General Conditions - Service Provider - Mobilization	TRUCK PER EVENT	$ 5,000	10	$ 50,000	10	$ 50,000	10	$ 50,000	1
1.3	**MAINTENANCE & MONITORING**				**$ 3,000**		**$ 4,000**		**$ 19,000**	1
1.3.1	Process equipment maintenance	% equipment	3%	1	$ 2,000	1	$ 3,000	1	$ 13,000	1
1.3.2	General building maintenance	sq foot	$ 2.50	50	$ 1,000	100	$ 1,000	2,100	$ 6,000	1
1.4	**ANALYTICAL MONITORING**				**$ 16,000**		**$ 36,000**		**$ 44,000**	1
1.4.1	Monitoring - Monthly samples	SAMPLE+SHIPPING	$ 400	40	$ 16,000	90	$ 36,000	110	$ 44,000	1
Item Num.	**Description of O&M Category**	**Unit of Measure**	**Annual Salary (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Notes**
2	**FIXED COSTS**									
2.1	**O&M LABOR**									
2.1.1	Water Treatment Operator	FTE	$ 100,000	0.5	$ 50,000	0.5	$ 50,000	1	$ 100,000	1
2.1.2	Shift Maintenance (Mechanical, Electrical)	FTE	$ 100,000	0.5	$ 50,000	0.5	$ 50,000	0.5	$ 50,000	1
	Total Workforce	**Base Payroll**			$ 100,000		$ 100,000		$ 150,000	1
		Fringe Rate	0.5		$ 50,000		$ 50,000		$ 75,000	1
		Total Payroll			**$ 150,000**		**$ 150,000**		**$ 225,000**	1
	ESTIMATED TOTAL ANNUAL OPERATING COSTS				**$ 220,000**		**$ 260,000**		**$ 520,000**	3
	Estimated Uncertainty Range	-30%			$ 160,000		$ 190,000		$ 370,000	3,4
		+50%			$ 330,000		$ 390,000		$ 780,000	3,4
	Unit Cost per 1,000 gallons treated				$ 419		$ 49		$ 9.89	

Notes

[1] Annual total cost values are rounded up to the nearest thousand USD.

[2] Assumed operating 7 days of operation per week, 365 days per year. Adjusted based on flow. Reference: Electric Power Research Institute and Water Research Foundation. Electricity Use and Management in the Municipal Water Supply and Wastewater Industries. Report 3002001433. November 2013.

[3] Values less than $1 million are rounded to the nearest ten thousand USD, and values greater than or equal to $1 million are rounded up to the nearest one hundred thousand USD.

[4] Operation and maintenance costs are based on a Class 5 capital cost estimate with a +50/-30% uncertainty as applicable for projects at less than 2% of full project definition per AACE International 17R-97. Operation and maintenance costs are also expected to have a +50/-30% uncertainty.

Appendix E

Alternative 6a (GAC, AIX, Incineration) – Municipal Wastewater

Appendix E: Table 6a - WW	**Alternative 6a - GAC, AIX, Incineration**			**Units**	**Notes**
Waste stream	Wastewater	Wastewater	Wastewater	--	
Design flow rate	0.1	1	10	MGD	
Design flow rate	70	700	7,000	gpm	Rounded to one significant digit.
Pressure Vessels (GAC)					
Vessel capacity	6,000	20,000	60,000	lb	
# of lead/lag vessel trains	1	3	9	--	
# of vessels	2	6	18	--	
Media type	12x40	12x40	12x40	--	Reagglomerated, coal-based GAC.
Media apparent density	0.54	0.54	0.54	g/cm3	
Media volume, per vessel	150	470	1,600	ft3	Calculated based on target EBCT. Rounded to two significant digits.
Media weight, per vessel	5,100	16,000	54,000	lb	Calculated based on target EBCT. Rounded to two significant digits.
Empty-bed contact time, per vessel	15	15	15	minutes	
Vessel diameter	6	8	14	ft	Showing two significant digits.
Media bed depth	5	9	10	ft	Showing two significant digits.
Hydraulic loading rate	2.5	4.6	5.1	gpm/ft2	Showing two significant digits.
Estimated footprint, first train	800	2,400	3,300	ft2	Rounded to the nearest 100 ft2.
Estimated footprint, additional train	600	1,300	2,100	ft2	Rounded to the nearest 100 ft2.
Driving lane	No	Yes	Yes	--	
Pipe gallery	No	Yes	Yes	--	
Lab and bathroom	No	Yes	Yes	--	
Estimated footprint, total (GAC)	800	5,000	20,100	ft2	Rounded to the nearest 100 ft2.
Pressure Vessels (Single Use AIX)					
Vessel capacity	38	201	450	ft3	
# of lead/lag vessel trains	2	2	9	--	
# of vessels	4	4	18	--	
Media type	AIX Resin	AIX Resin	AIX Resin	--	Polystyrenic gel.
Media apparent density	42.2	42.2	42.2	lb/ft3	

Appendix E: Table 6a - WW	**Alternative 6a - GAC, AIX, Incineration**			**Units**	**Notes**
Media volume, per vessel	19	190	420	ft3	Calculated based on target EBCT. Rounded to two significant digits.
Media weight, per vessel	900	8,100	18,000	lb	Calculated based on target EBCT. Rounded to two significant digits.
Empty-bed contact time, per vessel	4	4	4	minutes	
Vessel diameter	3	8	12	ft	
Media bed depth	3	4	4	ft	
Hydraulic loading rate	5.0	7.0	6.9	gpm/ft2	
Specific flowrate	0.9	1.7	1.7	gpm/ft3	
Estimated footprint, first train	500	900	1200	ft2	Rounded to the nearest 100 ft2.
Estimated footprint, additional train	400	700	1000	ft2	Rounded to the nearest 100 ft2.
Estimated footprint, total (AIX)	900	1,600	9,200	ft2	Rounded to the nearest 100 ft2.
Combined footprint, total	1,700	6,600	30,000	ft2	Rounded to the nearest 100 ft2.
Estimated time to breakthrough, GAC	10,000	10,000	10,000	bed volumes	Based on modelled breakthrough of PFBS at 5 ng/L.
Estimated changeout frequency, GAC	100	100	100	days	Rounded down to the nearest 10 days.
# Trucks per event (virgin GAC)	1	2	13	--	Rounded to nearest integer.
# Trucks per event (spent GAC)	1	3	25	--	Assumes spent media is 2x weight after use, requiring 2x trucks to remove than deliver. Rounded to nearest integer.
Change-out events per year	3.7	3.7	3.7	--	Rounded to two significant digits.
Estimated media usage rate, GAC	22,000	220,000	2,000,000	lbs/year	Rounded to two significant digits.
Estimated media usage rate, GAC	0.60	0.60	0.55	lbs/1000-gal	Pounds per 1,000 gallons treated.
Estimated time to breakthrough, AIX	140,000	140,000	140,000	bed volumes	Based on the maximum bed volumes treated in Ellis et al. (2022).
Estimated changeout frequency, AIX	380	380	380	days	Rounded down to the nearest 10 days.
# Trucks per event (virgin resin)	1	1	5	--	Rounded to nearest integer.

Appendix E: Table 6a - WW	Alternative 6a - GAC, AIX, Incineration			Units	Notes
# Trucks per event (spent resin)	1	1	9	--	Assumes spent media is 2x weight after use, requiring 2x trucks to remove than deliver. Rounded to nearest integer.
Change-out events per year	1	1	1	--	Rounded to one significant digit.
Estimated media usage rate, AIX	1,800	16,000	160,000	lb/year	Rounded to two significant digits.
Estimated media usage rate	0.049	0.044	0.044	lbs/1000-gal	Pounds per 1,000 gallons treated.

PREPARED BY: BARR ENGINEERING COMPANY

ENGINEER'S OPINION OF PROBABLE PROJECT COST

PROJECT: Evaluation of Current Alternatives and Estimated Costs for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water
LOCATION: MINNESOTA
PROJECT #: 23621432

OPINION OF COST - SUMMARY

SHEET:	1	OF	1
BY:	RRM	DATE:	12/8/2022
CHECKED BY:	AJM2	DATE:	3/3/2023
APPROVED BY:	KMB	DATE:	3/20/2023
ISSUED:	DRAFT 1	DATE:	11/11/2022
ISSUED:	DRAFT 2	DATE:	12/22/2022
ISSUED:	FINAL	DATE:	3/24/2023
ISSUED:		DATE:	

Engineer's Opinion of Probable Capital Cost
Municipal Wastewater - Alternative 6a

Total Treatment Flow Rate			Units
Low	Middle	High	
0.1	1	10	MGD
70	700	7000	GPM

			LOW FLOW RATE			MIDDLE FLOW RATE			HIGH FLOW RATE			
Cat. No.	**ITEM DESCRIPTION**	**UNIT**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**NOTES**
A	General Conditions	LS	1	$303,000	$303,000	1	$1,722,000	$1,722,000	1	$11,906,000	$11,906,000	1,2,3,4
B	PFAS Separation Equipment	LS	1	$442,000	$442,000	1	$3,203,000	$3,203,000	1	$25,661,000	$25,661,000	1,2,3,4
C	Piping and Appurtenances	LS	1	$221,000	$221,000	1	$1,602,000	$1,602,000	1	$12,831,000	$12,831,000	1,2,3,4
D	Electrical and Instrumentation/Controls	LS	1	$89,000	$89,000	1	$641,000	$641,000	1	$5,133,000	$5,133,000	1,2,3,4
E	Treatment Building	SF	1,700	$500	$850,000	6,600	$500	$3,300,000	30,000	$500	$15,000,000	1,2,3,4
F	Site Work	LS	1	$85,000	$85,000	1	$330,000	$330,000	1	$1,500,000	$1,500,000	1,2,3,4
G	Installation (Equipment and Piping)	LS	1	$332,000	$332,000	1	$2,403,000	$2,403,000	1	$19,246,000	$19,246,000	1,2,3,4
	CONSTRUCTION SUBTOTAL				$2,320,000			$13,200,000			$91,280,000	1,2,3,4,5
	CONSTRUCTION CONTINGENCY			25%	$580,000		25%	$3,300,000		25%	$22,820,000	1,4,5
	ESTIMATED CONSTRUCTION COST				$2,900,000			$16,500,000			$114,100,000	1,2,3,4,5
	ESTIMATED TOTAL PROJECT COST				**$2,900,000**			**$16,500,000**			**$114,100,000**	1,2,3,4,6
	ESTIMATED ACCURACY RANGE	-30%			$2,100,000			$11,600,000			$79,900,000	4,6
		50%			$4,400,000			$24,800,000			$171,200,000	4,6

Notes

[1] Limited design work completed.

[2] Quantities based on design work completed.

[3] Unit prices based on information available at this time.

[4] This feasibility-level (Class 5, 0-2% design completion per AACE International Recommended Practice No. 17R-97) cost estimate is based on preliminary designs, quantities and unit prices. Costs will change with further design. Time value-of-money escalation costs are not included. Contingency is an allowance for the net sum of costs that will be in the Final Total Project Cost at the time of the completion of design, but are not included at this level of project definition. The estimated accuracy range for the Total Project Cost as the project is defined is -30% to +50%. The accuracy range is based on professional judgement considering the level of design completed, the complexity of the project and the uncertainties in the project as scoped. The contingency and the accuracy range are not intended to include costs for future scope changes that are not part of the project as currently scoped or costs for risk contingency. Operation and maintenance costs are not included.

[5] Estimate costs are reported to nearest ten thousand dollars.

[6] Estimate costs are rounded up to nearest one hundred thousand dollars. Estimate costs are to construct each alternative. The estimated costs do not include planning, engineering and design, permitting, construction administration, maintenance, monitoring or additional tasks following construction.

BARR

PREPARED BY: BARR ENGINEERING COMPANY

ENGINEER'S OPINION OF PROBABLE PROJECT COST

PROJECT: Evaluation of Current Alternatives and Estimated Costs for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water
LOCATION: MINNESOTA
PROJECT #: 23621432
OPINION OF COST - SUMMARY

SHEET:	1	OF	1
BY:	AJM2	DATE:	12/9/2022
CHECKED BY:	RRM	DATE:	2/17/2023
APPROVED BY:	KMB	DATE:	3/20/2023

ISSUED:	DRAFT 1	DATE:	11/11/2022
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ISSUED:	FINAL	DATE:	3/24/2023
ISSUED:		DATE:	

Engineer's Opinion of Probable O&M Cost

Municipal Wastewater - Alternative 6a - GAC+AIX

Total Treatment Flow Rate			Units
Low	**Middle**	**High**	
0.1	1	10	MGD
70	700	7000	GPM

				LOW FLOWRATE		**MIDDLE FLOWRATE**		**HIGH FLOWRATE**		
Item Num.	**Description of O&M Category**	**Unit of Measure**	**Unit Rate (USD/unit)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Notes**
1	**VARIABLE COSTS**									
1.1	**UTILITIES**				**$ 12,000**		**$ 27,000**		**$ 161,000**	1
1.1.1	Electricity - Process equipment	kWh	$ 0.13	11,700	$ 2,000	43,500	$ 6,000	361,700	$ 49,000	1,2
1.1.2	Electricity - Non-process energy load (building power, HVAC, etc.)	kWh	$ 0.13	63,000	$ 9,000	121,800	$ 17,000	710,400	$ 95,000	1,2
1.1.3	Natural gas - Non-process energy load (heating)	MMBTU	$ 15	60	$ 1,000	230	$ 4,000	1,100	$ 17,000	1
1.2	**MEDIA REPLACEMENT AND DISPOSAL**				**$ 151,000**		**$ 1,140,000**		**$ 6,431,000**	1
1.2.1	New Media Purchase - GAC	LB	$ 2.50	22,200	$ 56,000	222,000	$ 555,000	0	$ -	1
1.2.2	Transportation - New Media Purchase - GAC	MILE	$ 3.00	3,300	$ 10,000	6,700	$ 21,000	0	$ -	1
1.2.3	Media disposal - Incineration - GAC	LB	$ 0.65	44,400	$ 29,000	444,000	$ 289,000	0	$ -	1
1.2.4	Transportation - Media incineration - GAC	MILE	$ 3.00	2,800	$ 9,000	8,400	$ 26,000	0	$ -	1
1.2.5	New Media Purchase - AIX	CU FT	$ 320	40	$ 13,000	380	$ 122,000	3,800	$ 1,216,000	1
1.2.6	Transportation - New Media Purchase - AIX	MILE	$ 3.00	900	$ 3,000	900	$ 3,000	4,500	$ 14,000	1
1.2.7	Media disposal - Incineration - AIX	LB	$ 0.65	3,400	$ 3,000	32,100	$ 21,000	320,700	$ 209,000	1
1.2.8	Transportation - Media incineration - AIX	MILE	$ 3.00	760	$ 3,000	760	$ 3,000	6,800	$ 21,000	1
1.2.9	Media disposal - Reactivation (Destruction + Media)	LB	$ 2.00	0	$ -	0	$ -	1,998,000	$ 3,996,000	1
1.2.10	Transportation - Media reactivation	MILE	$ 3.00	0	$ -	0	$ -	166,500	$ 500,000	1
1.2.11	General Conditions - Service Provider - Mobilization - GAC	TRUCK PER EVENT	$ 5,000	5	$ 25,000	20	$ 100,000	95	$ 475,000	1
1.2.12	General Conditions - Service Provider - Mobilization - AIX	TRUCK PER EVENT	$ 5,000	2	$ 10,000	2	$ 10,000	15	$ 75,000	1
1.3	**MAINTENANCE & MONITORING**				**$ 28,000**		**$ 181,000**		**$ 1,384,000**	1
1.3.1	Process equipment maintenance	% equipment	3%	1	$ 23,000	1	$ 164,000	1	$ 1,309,000	1
1.3.2	General building maintenance	sq foot	$ 2.50	1,700	$ 5,000	6,600	$ 17,000	30,000	$ 75,000	1
1.4	**ANALYTICAL MONITORING**				**$ 40,000**		**$ 60,000**		**$ 184,000**	1
1.4.1	Monitoring - Monthly samples	SAMPLE+SHIPPING	$ 400	100	$ 40,000	150	$ 60,000	460	$ 184,000	1

Item Num.	**Description of O&M Category**	**Unit of Measure**	**Annual Salary (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Notes**
2	**FIXED COSTS**									
2.1	**O&M LABOR**									
2.1.1	Water Treatment Operator	FTE	$ 100,000	1	$ 100,000	1	$ 100,000	2	$ 200,000	1
2.1.2	Shift Maintenance (Mechanical, Electrical)	FTE	$ 100,000	0.5	$ 50,000	0.5	$ 50,000	1	$ 100,000	1
	Total Workforce	**Base Payroll**			$ 150,000		$ 150,000		$ 300,000	1
		Fringe Rate	0.5		$ 75,000		$ 75,000		$ 150,000	1
		Total Payroll			**$ 225,000**		**$ 225,000**		**$ 450,000**	1
	ESTIMATED TOTAL ANNUAL OPERATING COSTS				**$ 460,000**		**$ 1,700,000**		**$ 8,700,000**	3
	Estimated Uncertainty Range	-30%			$ 330,000		$ 1,190,000		$ 6,100,000	3,4
		+50%			$ 690,000		$ 2,550,000		$ 13,100,000	3,4
	Unit Cost per 1,000 gallons treated				$ 12.60		$ 4.66		$ 2.38	

Notes

[1] Annual total cost values are rounded up to the nearest thousand USD.

[2] Assumed operating 7 days of operation per week, 365 days per year. Adjusted based on flow. Reference: Electric Power Research Institute and Water Research Foundation. Electricity Use and Management in the Municipal Water Supply and Wastewater Industries. Report 3002001433. November 2013.

[3] Values less than $1 million are rounded to the nearest ten thousand USD, and values greater than or equal to $1 million are rounded up to the nearest one hundred thousand USD.

[4] Operation and maintenance costs are based on a Class 5 capital cost estimate with a +50/-30% uncertainty as applicable for projects at less than 2% of full project definition per AACE International 17R-97. Operation and maintenance costs are also expected to have a +50/-30% uncertainty.

Appendix E

Alternative 6b (RO, GAC, AIX, Incineration) – Municipal Wastewater

Appendix E: Table 6b - WW	**Alternative 6b - RO, Conc. to GAC and AIX, Incineration**			**Units**	**Notes**
Waste stream	Wastewater	Wastewater	Wastewater	--	
Design flow rate	0.1	1	10	MGD	
Design flow rate	70	700	7,000	gpm	Rounded to one significant digit.
Membranes					
Recovery	85%	85%	85%	%	Assumed.
# of skids	1	1	7	--	
Membrane type	RO	RO	RO	--	
# Elements	18	156	1,554	--	
Flux	16	16	16	gfd	gfd = gallons per sq. ft. per day.
Concentrate management flow rate	0.02	0.2	1.5	MGD	
Concentrate management flow rate	20	110	1,100	gpm	Rounded to two significant digits.
Estimated footprint, total (membranes)	600	700	3,500	ft2	
Pressure Vessels (GAC)					
Vessel capacity	750	6,000	20,000	lb	
# of lead/lag vessel trains	2	2	4	--	
# of vessels	4	4	8	--	
Media type	12x40	12x40	12x40	--	Reagglomerated, coal-based GAC.
Media apparent density	0.54	0.54	0.54	g/cm3	
Media volume, per vessel	21	120	560	ft3	Calculated based on target EBCT. Rounded to two significant digits.
Media weight, per vessel	710	4,100	19,000	lb	Calculated based on target EBCT. Rounded to two significant digits.
Empty-bed contact time, per vessel	15	15	15	minutes	
Vessel diameter	2.0	6.0	12	ft	Showing two significant digits.
Media bed depth	6.4	3.9	4.9	ft	Showing two significant digits.
Hydraulic loading rate	3.2	1.9	2.4	gpm/ft2	Showing two significant digits.
Estimated footprint, first train	200	2,200	3000	ft2	Rounded to the nearest 100 ft2.
Estimated footprint, additional train	100	1,000	1800	ft2	Rounded to the nearest 100 ft2.
Driving lane	No	Yes	Yes	--	

Appendix E: Table 6b - WW	**Alternative 6b - RO, Conc. to GAC and AIX, Incineration**			**Units**	**Notes**
Pipe gallery	No	Yes	Yes	--	
Lab and bathroom	No	Yes	Yes	--	
Estimated footprint, total (GAC)	300	3,200	8,400	ft2	Rounded to the nearest 100 ft2.
Pressure Vessels (AIX)					
Vessel capacity	5	38	201	ft3	
# of lead/lag vessel trains	3	2	3	--	
# of vessels	6	4	6	--	
Media type	AIX Resin	AIX Resin	AIX Resin	--	Polystyrenic gel.
Media apparent density	42.2	42.2	42.2	lb/ft3	
Media volume, per vessel	4	30	200	ft3	Calculated based on target EBCT. Rounded to one significant digit.
Media weight, per vessel	170	1,300	8,500	lb	Calculated based on target EBCT. Rounded to two significant digits.
Empty-bed contact time, per vessel	4	4	4	minutes	
Vessel diameter	1.5	3	8	ft	
Media bed depth	2	4	4	ft	
Hydraulic loading rate	3.8	7.8	7.3	gpm/ft2	
Specific flowrate	1.3	1.4	1.8	gpm/ft3	
Estimated footprint, first train	50	200	900	ft2	70 gpm rounded to the nearest 10 ft2. 700 and 7,000 gpm rounded to the nearest 100 ft2.
Estimated footprint, additional train	40	200	700	ft2	70 gpm rounded to the nearest 10 ft2. 700 and 7,000 gpm rounded to the nearest 100 ft2.
Estimated footprint, total	130	400	2,300	ft2	70 gpm rounded to the nearest 10 ft2. 700 and 7,000 gpm rounded to the nearest 100 ft2.
Combined footprint, total	1,100	4,300	14,200	ft2	Rounded to the nearest 100 ft2.

Appendix E: Table 6b - WW	**Alternative 6b - RO, Conc. to GAC and AIX, Incineration**			**Units**	**Notes**
Estimated time to breakthrough, GAC	8,100	8,100	8,100	bed volumes	Based on the modelled breakthrough of PFBS at 5 ng/L.
Estimated changeout frequency, GAC	80	80	80	days	Rounded down to the nearest 10 days.
# Trucks per event (dry GAC)	1	1	2	--	Rounded to nearest integer.
# Trucks per event (wet GAC)	1	1	4	--	Assumes spent media is 2x weight after use, requiring 2x trucks to remove than deliver. Rounded to nearest integer.
Change-out events per year	4.6	4.6	4.6	--	Rounded to two significant digits.
Estimated media usage rate, GAC	6,900	55,000	370,000	lbs/year	Rounded to two significant digits.
Estimated media usage rate, GAC	0.19	0.151	0.101	lbs/1000-gal	Pounds per 1,000 gallons treated.
Estimated time to breakthrough, AIX	140,000	140,000	140,000	bed volumes	Based on the maximum bed volumes treated in Ellis et al. (2022).
Estimated changeout frequency, AIX	380	380	380	days	Rounded down to the nearest 10 days.
# Trucks per event (virgin resin)	1	1	1	--	Rounded to nearest integer.
# Trucks per event (spent resin)	1	1	2	--	Assumes spent media is 2x weight after use, requiring 2x trucks to remove than deliver. Rounded to nearest integer.
Change-out events per year	1	1	1	--	Rounded to one significant digit.
Estimated media usage rate, AIX	490	2,500	25,000	lb/year	Rounded to two significant digits.
Estimated media usage rate	0.013	0.007	0.007	lbs/1000-gal	Pounds per 1,000 gallons treated.

PREPARED BY: BARR ENGINEERING COMPANY

ENGINEER'S OPINION OF PROBABLE PROJECT COST

PROJECT: Evaluation of Current Alternatives and Estimated Costs for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water
LOCATION: MINNESOTA
PROJECT #: 23621432

OPINION OF COST - SUMMARY

SHEET:	1	OF	1
BY:	RRM	DATE:	12/8/2022
CHECKED BY:	AJM2	DATE:	3/3/2023
APPROVED BY:	KMB	DATE:	3/20/2023
ISSUED:	DRAFT 1	DATE:	11/11/2022
ISSUED:	DRAFT 2	DATE:	12/22/2022
ISSUED:	FINAL	DATE:	3/24/2023
ISSUED:		DATE:	

Engineer's Opinion of Probable Capital Cost
Municipal Wastewater - Alternative 6b

Total Treatment Flow Rate			Units
Low	Medium	High	
0.1	1	10	MGD
70	700	7000	GPM

			LOW FLOW RATE			MIDDLE FLOW RATE			HIGH FLOW RATE			
Cat. No.	**ITEM DESCRIPTION**	**UNIT**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**NOTES**
A	General Conditions	LS	1	$263,000	$263,000	1	$1,018,000	$1,018,000	1	$9,819,000	$9,819,000	1,2,3,4
B	PFAS Separation Equipment	LS	1	$467,000	$467,000	1	$1,804,000	$1,804,000	1	$23,530,000	$23,530,000	1,2,3,4
C	Piping and Appurtenances	LS	1	$234,000	$234,000	1	$902,000	$902,000	1	$11,765,000	$11,765,000	1,2,3,4
D	Electrical and Instrumentation/Controls	LS	1	$94,000	$94,000	1	$361,000	$361,000	1	$4,706,000	$4,706,000	1,2,3,4
E	Treatment Building	SF	1,100	$500	$550,000	4,300	$500	$2,150,000	14,200	$500	$7,100,000	1,2,3,4
F	Site Work	LS	1	$55,000	$55,000	1	$215,000	$215,000	1	$710,000	$710,000	1,2,3,4
G	Installation (Equipment and Piping)	LS	1	$351,000	$351,000	1	$1,353,000	$1,353,000	1	$17,648,000	$17,648,000	1,2,3,4
	CONSTRUCTION SUBTOTAL				$2,010,000			$7,800,000			$75,280,000	1,2,3,4,5
	CONSTRUCTION CONTINGENCY			25%	$500,000		25%	$1,950,000		25%	$18,820,000	1,4,5
	ESTIMATED CONSTRUCTION COST				$2,510,000			$9,750,000			$94,100,000	1,2,3,4,5
	ESTIMATED TOTAL PROJECT COST				**$2,600,000**			**$9,800,000**			**$94,100,000**	1,2,3,4,6
	ESTIMATED ACCURACY RANGE	-30%			$1,900,000			$6,900,000			$65,900,000	4,6
		50%			$3,900,000			$14,700,000			$141,200,000	4,6

Notes

[1] Limited design work completed.

[2] Quantities based on design work completed.

[3] Unit prices based on information available at this time.

[4] This feasibility-level (Class 5, 0-2% design completion per AACE International Recommended Practice No. 17R-97) cost estimate is based on preliminary designs, quantities and unit prices. Costs will change with further design. Time value-of-money escalation costs are not included. Contingency is an allowance for the net sum of costs that will be in the Final Total Project Cost at the time of the completion of design, but are not included at this level of project definition. The estimated accuracy range for the Total Project Cost as the project is defined is -30% to +50%. The accuracy range is based on professional judgement considering the level of design completed, the complexity of the project and the uncertainties in the project as scoped. The contingency and the accuracy range are not intended to include costs for future scope changes that are not part of the project as currently scoped or costs for risk contingency. Operation and maintenance costs are not included.

[5] Estimate costs are reported to nearest ten thousand dollars.

[6] Estimate costs are rounded up to nearest one hundred thousand dollars. Estimate costs are to construct each alternative. The estimated costs do not include planning, engineering and design, permitting, construction administration, maintenance, monitoring or additional tasks following construction.

PREPARED BY: BARR ENGINEERING COMPANY

ENGINEER'S OPINION OF PROBABLE PROJECT COST

PROJECT: Evaluation of Current Alternatives and Estimated Costs for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water
LOCATION: MINNESOTA
PROJECT #: 23621432
OPINION OF COST - SUMMARY

SHEET:	1	OF	1
BY:	AJM2	DATE:	12/9/2022
CHECKED BY:	RRM	DATE:	2/17/2023
APPROVED BY:	KMB	DATE:	3/20/2023
ISSUED:	DRAFT 1	DATE:	11/11/2022
ISSUED:	DRAFT 2	DATE:	12/22/2022
ISSUED:	FINAL	DATE:	3/24/2023
ISSUED:		DATE:	

Engineer's Opinion of Probable O&M Cost

Municipal Wastewater - Alternative 6b - RO+GAC+AIX

Total Treatment Flow Rate			Units
Low	Middle	High	
0.1	1	10	MGD
70	700	7000	GPM

				LOW FLOWRATE		MIDDLE FLOWRATE		HIGH FLOWRATE		
Item Num.	**Description of O&M Category**	**Unit of Measure**	**Unit Rate (USD/unit)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Notes**
1	**VARIABLE COSTS**									
1.1	**UTILITIES**				**$ 34,000**		**$ 249,000**		**$ 2,381,000**	1
1.1.1	Electricity - Process equipment	kWh	$ 0.13	179,200	$ 24,000	1,719,000	$ 229,000	17,113,000	$ 2,278,000	1,2
1.1.2	Electricity - Non-process energy load (building power, HVAC, etc.)	kWh	$ 0.13	63,000	$ 9,000	121,800	$ 17,000	710,400	$ 95,000	1,2
1.1.3	Natural gas - Non-process energy load (heating)	MMBTU	$ 15	40	$ 1,000	150	$ 3,000	500	$ 8,000	1
1.2	**MEDIA REPLACEMENT AND DISPOSAL**				**$ 112,000**		**$ 314,000**		**$ 1,994,000**	1
1.2.1	New Media Purchase - GAC	LB	$ 2.50	6,900	$ 18,000	55,200	$ 138,000	0	$ -	1
1.2.2	Transportation - New Media Purchase - GAC	MILE	$ 3.00	4,100	$ 13,000	4,100	$ 13,000	0	$ -	1
1.2.3	Media disposal - Incineration - GAC	LB	$ 0.65	13,800	$ 9,000	110,400	$ 72,000	0	$ -	1
1.2.4	Transportation - Media incineration - GAC	MILE	$ 3.00	3,500	$ 11,000	3,500	$ 11,000	0	$ -	1
1.2.5	New Media Purchase - AIX	CU FT	$ 320	10	$ 4,000	60	$ 20,000	2,800	$ 896,000	1
1.2.6	Transportation - New Media Purchase - AIX	MILE	$ 3.00	900	$ 3,000	900	$ 3,000	900	$ 3,000	1
1.2.7	Media disposal - Incineration - AIX	LB	$ 0.65	840	$ 1,000	5,100	$ 4,000	236,300	$ 154,000	1
1.2.8	Transportation - Media incineration - AIX	MILE	$ 3.00	760	$ 3,000	760	$ 3,000	1,500	$ 5,000	1
1.2.9	Media disposal - Reactivation (Destruction + Media)	LB	$ 2.00	0	$ -	0	$ -	368,000	$ 736,000	1
1.2.10	Transportation - Media reactivation	MILE	$ 3.00	0	$ -	0	$ -	33,100	$ 100,000	1
1.2.11	General Conditions - Service Provider - Mobilization - GAC	TRUCK PER EVENT	$ 5,000	10	$ 50,000	10	$ 50,000	20	$ 100,000	1
1.2.12	General Conditions - Service Provider - Mobilization - AIX	TRUCK PER EVENT	$ 5,000	2	$ 10,000	2	$ 10,000	5	$ 25,000	1
1.3	**RO CONSUMABLES**				**$ 55,000**		**$ 463,000**		**$ 4,617,000**	1
1.3.1	Antiscalant	LB	$ 10	670	$ 7,000	6,700	$ 67,000	67,000	$ 670,000	1
1.3.2	Clean-in-place Chemicals	LB	$ 2	21,600	$ 44,000	187,500	$ 375,000	1,867,000	$ 3,730,000	1
1.3.3	RO Element Replacement (5-year Replacement)	Element	$ 700	5	$ 4,000	30	$ 21,000	310	$ 217,000	1
1.4	**MAINTENANCE & MONITORING**				**$ 27,000**		**$ 104,000**		**$ 1,237,000**	1
1.4.1	Process equipment maintenance	% equipment	3%	1	$ 24,000	1	$ 93,000	1	$ 1,201,000	1
1.4.2	General building maintenance	sq foot	$ 2.50	1,100	$ 3,000	4,300	$ 11,000	14,200	$ 36,000	1
1.5	**ANALYTICAL MONITORING**				**$ 68,000**		**$ 60,000**		**$ 88,000**	1
1.5.1	Monitoring - Monthly samples	SAMPLE+SHIPPING	$ 400	170	$ 68,000	150	$ 60,000	220	$ 88,000	1
Item Num.	**Description of O&M Category**	**Unit of Measure**	**Annual Salary (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Notes**
2	**FIXED COSTS**									
2.1	**O&M LABOR**									
2.1.1	Water Treatment Operator	FTE	$ 100,000	2	$ 200,000	2	$ 200,000	4	$ 400,000	1
2.1.2	Shift Maintenance (Mechanical, Electrical)	FTE	$ 100,000	1	$ 100,000	1	$ 100,000	2	$ 200,000	1
	Total Workforce	**Base Payroll**			$ 300,000		$ 300,000		$ 600,000	1
		Fringe Rate	0.5		$ 150,000		$ 150,000		$ 300,000	1
		Total Payroll			**$ 450,000**		**$ 450,000**		**$ 900,000**	1
	ESTIMATED TOTAL ANNUAL OPERATING COSTS				**$ 750,000**		**$ 1,700,000**		**$ 11,300,000**	3
	Estimated Uncertainty Range	-30%			$ 530,000		$ 1,190,000		$ 8,000,000	3,4
		+50%			$ 1,130,000		$ 2,550,000		$ 16,950,000	3,4
	Unit Cost per 1,000 gallons treated				$ 20.55		$ 4.66		$ 3.10	

Notes

[1] Annual total cost values are rounded up to the nearest thousand USD.

[2] Assumed operating 7 days of operation per week, 365 days per year. Adjusted based on flow. Reference: Electric Power Research Institute and Water Research Foundation. Electricity Use and Management in the Municipal Water Supply and Wastewater Industries. Report 3002001433. November 2013.

[3] Values less than $1 million are rounded to the nearest ten thousand USD, and values greater than or equal to $1 million are rounded up to the nearest one hundred thousand USD.

[4] Operation and maintenance costs are based on a Class 5 capital cost estimate with a +50/-30% uncertainty as applicable for projects at less than 2% of full project definition per AACE International 17R-97. Operation and maintenance costs are also expected to have a +50/-30% uncertainty.

Appendix E

Alternative 8a (Foam Fractionation, Incineration) – Landfill Leachate / Compost Contact Water

Appendix E: Table 8a - LL/CCW	**Alternative 8a - Foam Fractionation w/ Incineration**			**Units**	**Notes**
Waste stream	LL/CCW	LL/CCW	LL/CCW	--	
Design flow rate	0.0014	0.014	0.14	MGD	
Design flow rate	1,440	14,400	144,000	GPD	
Design flow rate	1	10	100	gpm	
Expected foamate flow rate	0.0014	0.014	0.144	GPD	
Expected foamate flow rate	0.000001	0.00001	0.0001	gpm	
Foam Fractionation Equipment					
# of package systems	1	1	1	--	
Primary fractionation vessels	2	2	4	--	
Secondary fractionation vessels	1	1	2	--	
Lab and bathroom	No	No	No		
Estimated footprint, total	1,000	1,000	1,000	ft2	Rounded to the nearest 100 ft2.

PREPARED BY: BARR ENGINEERING COMPANY

ENGINEER'S OPINION OF PROBABLE PROJECT COST

PROJECT: Evaluation of Current Alternatives and Estimated Costs for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water
LOCATION: MINNESOTA
PROJECT #: 23621432

OPINION OF COST - SUMMARY

SHEET:	1	OF	1
BY:	KMB	DATE:	10/10/2022
CHECKED BY:	AJM2	DATE:	3/3/2023
APPROVED BY:	KMB	DATE:	3/20/2023
ISSUED:	DRAFT 1	DATE:	11/11/2022
ISSUED:	DRAFT 2	DATE:	12/22/2022
ISSUED:	FINAL	DATE:	3/24/2023
ISSUED:		DATE:	

Engineer's Opinion of Probable Capital Cost Landfill Leachate / Compost Contact Water - Alternative 8a

Total Treatment Flow Rate			Units
Low	Middle	High	
0.00144	0.0144	0.144	MGD
1	10	100	GPM

			LOW FLOW RATE			MIDDLE FLOW RATE			HIGH FLOW RATE			
Cat. No.	ITEM DESCRIPTION	UNIT	QUANTITY	UNIT COST	ITEM COST	QUANTITY	UNIT COST	ITEM COST	QUANTITY	UNIT COST	ITEM COST	NOTES
A	General Conditions	LS	1	$742,000	$742,000	1	$742,000	$742,000	1	$883,000	$883,000	1,2,3,4
B	PFAS Separation Equipment	LS	1	$1,792,000	$1,792,000	1	$1,792,000	$1,792,000	1	$2,176,000	$2,176,000	1,2,3,4
C	Piping and Appurtenances	LS	1	$896,000	$896,000	1	$896,000	$896,000	1	$1,088,000	$1,088,000	1,2,3,4
D	Electrical and Instrumentation/Controls	LS	1	$359,000	$359,000	1	$359,000	$359,000	1	$436,000	$436,000	1,2,3,4
E	Treatment Building	SF	1,000	$500	$500,000	1,000	$500	$500,000	1,000	$500	$500,000	1,2,3,4
F	Site Work	LS	1	$50,000	$50,000	1	$50,000	$50,000	1	$50,000	$50,000	1,2,3,4
G	Installation (Equipment and Piping)	LS	1	$1,344,000	$1,344,000	1	$1,344,000	$1,344,000	1	$1,632,000	$1,632,000	1,2,3,4
	CONSTRUCTION SUBTOTAL				$5,680,000			$5,680,000			$6,770,000	1,2,3,4,5
	CONSTRUCTION CONTINGENCY			25%	$1,420,000		25%	$1,420,000		25%	$1,690,000	1,4,5
	ESTIMATED CONSTRUCTION COST				$7,100,000			$7,100,000			$8,460,000	1,2,3,4,5
	ESTIMATED TOTAL PROJECT COST				**$7,100,000**			**$7,100,000**			**$8,500,000**	1,2,3,4,6
	ESTIMATED ACCURACY RANGE	-30%			$5,000,000			$5,000,000			$6,000,000	4,6
		50%			$10,700,000			$10,700,000			$12,800,000	4,6

Notes

[1] Limited design work completed.

[2] Quantities based on design work completed.

[3] Unit prices based on information available at this time.

[4] This feasibility-level (Class 5, 0-2% design completion per AACE International Recommended Practice No. 17R-97) cost estimate is based on preliminary designs, quantities and unit prices. Costs will change with further design. Time value-of-money escalation costs are not included. Contingency is an allowance for the net sum of costs that will be in the Final Total Project Cost at the time of the completion of design, but are not included at this level of project definition. The estimated accuracy range for the Total Project Cost as the project is defined is -30% to +50%. The accuracy range is based on professional judgement considering the level of design completed, the complexity of the project and the uncertainties in the project as scoped. The contingency and the accuracy range are not intended to include costs for future scope changes that are not part of the project as currently scoped or costs for risk contingency. Operation and maintenance costs are not included.

[5] Estimate costs are reported to nearest ten thousand dollars.

[6] Estimate costs are rounded up to nearest one hundred thousand dollars. Estimate costs are to construct each alternative. The estimated costs do not include planning, engineering and design, permitting, construction administration, maintenance, monitoring or

PREPARED BY: BARR ENGINEERING COMPANY

SHEET:	1	OF	1
BY:	AJM2	DATE:	12/9/2022
CHECKED BY:	RRM	DATE:	2/17/2023
APPROVED BY:	KMB	DATE:	3/20/2023

ENGINEER'S OPINION OF PROBABLE PROJECT COST

PROJECT: Evaluation of Current Alternatives and Estimated Costs for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water
LOCATION: MINNESOTA
PROJECT #: 23621432
OPINION OF COST - SUMMARY

ISSUED:	DRAFT 1	DATE:	11/11/2022
ISSUED:	DRAFT 2	DATE:	12/22/2022
ISSUED:	FINAL	DATE:	3/24/2023
ISSUED:		DATE:	

Engineer's Opinion of Probable O&M Cost

Landfill Leachate & Compost Contact Water - Alternative 8a - Foam Fractionation

Total Treatment Flow Rate			Units
Low	Middle	High	
0.00144	0.0144	0.144	MGD
1	10	100	GPM

				LOW FLOWRATE		MIDDLE FLOWRATE		HIGH FLOWRATE		
Item Num.	**Description of O&M Category**	**Unit of Measure**	**Unit Rate (USD/unit)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Notes**
1	**VARIABLE COSTS**									
1.1	**UTILITIES**				**$ 2,000**		**$ 4,000**		**$ 23,000**	1
1.1.1	Electricity - Combined process and non-process energy load	kWh	$ 0.13	2,000	$ 1,000	16,000	$ 3,000	159,000	$ 22,000	1,2
1.1.2	Natural gas - Non-process energy load (heating)	MMBTU	$ 15	40	$ 1,000	40	$ 1,000	40	$ 1,000	1
1.2	**CONCENTRATE AND RESIDUAL DISPOSAL**				**$ 9,000**		**$ 9,000**		**$ 9,000**	1
1.2.1	Concentrate and residual disposal - Incineration	LB	$ 0.65	10	$ 1,000	50	$ 1,000	440	$ 1,000	1
1.2.2	Transportation - Concentrate or residual disposal	MILE	$ 3.00	760	$ 3,000	760	$ 3,000	760	$ 3,000	1
1.2.3	General Conditions - Service Provider - Mobilization	TRUCK PER EVENT	$ 5,000	1	$ 5,000	1	$ 5,000	1	$ 5,000	1
1.3	**MISC CONSUMABLES**				**$ 5,000**		**$ 5,000**		**$ 9,000**	1
1.3.1	Bag/Cartridge Filters	50-ct	$ 750	5	$ 4,000	5	$ 4,000	5	$ 4,000	1
1.3.2	Chemical Feed	LB	$ 5	10	$ 1,000	90	$ 1,000	900	$ 5,000	1
1.4	**MAINTENANCE**				**$ 95,000**		**$ 95,000**		**$ 114,000**	1
1.4.1	Process equipment maintenance	% equipment	3%	1	$ 92,000	1	$ 92,000	1	$ 111,000	1
1.4.2	General building maintenance	sq foot	$ 2.50	1,000	$ 3,000	1,000	$ 3,000	1,000	$ 3,000	1
1.5	**ANALYTICAL MONITORING**				**$ 16,000**		**$ 16,000**		**$ 16,000**	1
1.5.1	Monitoring - Monthly samples	SAMPLE+SHIPPING	$ 400	40	$ 16,000	40	$ 16,000	40	$ 16,000	1
Item Num.	**Description of O&M Category**	**Unit of Measure**	**Annual Salary (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Notes**
2	**FIXED COSTS**									
2.1	**O&M LABOR**									
2.1.1	Water Treatment Operator	FTE	$ 100,000	0.5	$ 50,000	0.5	$ 50,000	0.5	$ 50,000	1
2.1.2	Shift Maintenance (Mechanical, Electrical)	FTE	$ 100,000	0.5	$ 50,000	0.5	$ 50,000	0.5	$ 50,000	1
	Total Workforce	**Base Payroll**			$ 100,000		$ 100,000		$ 100,000	1
		Fringe Rate	0.5		$ 50,000		$ 50,000		$ 50,000	1
		Total Payroll			**$ 150,000**		**$ 150,000**		**$ 150,000**	1
	ESTIMATED TOTAL ANNUAL OPERATING COSTS				**$ 280,000**		**$ 280,000**		**$ 330,000**	3
	Estimated Uncertainty Range	-30%			$ 200,000		$ 200,000		$ 240,000	3,4
		+50%			$ 420,000		$ 420,000		$ 500,000	3,4
	Unit Cost per 1,000 gallons treated				$ 533		$ 53		$ 6.28	

Notes

[1] Annual total cost values are rounded up to the nearest thousand USD.

[2] Assumed operating 7 days of operation per week, 365 days per year. Adjusted based on flow. Reference: Burns, D. J., Hinrichsen, H. M., Stevenson, P., & Murphy, P. J. C. (2022). Commercial-scale remediation of per- and polyfluoroalkyl substances from a landfill leachate catchment using Surface-Active Foam Fractionation (SAFF). Remediation Journal, 32, 139–150.

[3] Values less than $1 million are rounded to the nearest ten thousand USD, and values greater than or equal to $1 million are rounded up to the nearest one hundred thousand USD.

[4] Operation and maintenance costs are based on a Class 5 capital cost estimate with a +50/-30% uncertainty as applicable for projects at less than 2% of full project definition per AACE International 17R-97. Operation and maintenance costs are also expected to have a +50/-30% uncertainty.

Appendix E

Pyrolysis / Gasification – Municipal Wastewater Biosolids

Appendix E: Table 1 Solids	Alternative Solids - Pyrolysis/Gasification			Units	Notes
Waste stream	Solids	Solids	Solids	--	
Design flow rate	4	9	15	MGD	Biosolids industry standard: 1 mgd of influent flow produces 6 -7 dry ton/day of digested sludge.
Wet solids production rate	8	21	36	wet metric tons/day	25% total solids.
Dry solids production rate	2	6	10	dry US tons/day	
Number of dryers	1	1	1	--	
Type of dryer	Paddle	Paddle	Belt		Dried product is 90% TS from each dryer.
Evaporation rate	560	1400	2400	lb water/hr	
Pyrolysis unit capacity	3	3	10	dry US tons/day	
Number of pyrolysis units	1	2	1	units	
Dryer footprint	3,600	4,800	9,600	ft2	
Pyrolysis footprint	200	400	400	ft2	
Lab and bathroom	Yes	Yes	Yes	--	
Estimated footprint, total	5,300	7,000	13,000	ft2	Rounded to the nearest 100 ft2.

PREPARED BY: HAZEN on behalf of
BARR ENGINEERING COMPANY

ENGINEER'S OPINION OF PROBABLE PROJECT COST

PROJECT: Evaluation of Current Alternatives and Estimated Costs for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water
LOCATION: MINNESOTA
PROJECT #: 23621432

OPINION OF COST - SUMMARY

SHEET:	1	OF	1
BY:	HAZEN	DATE:	10/10/2022
CHECKED BY:	ALL	DATE:	12/12/2022
APPROVED BY:	HAZEN	DATE:	3/15/2023
ISSUED:	DRAFT 1	DATE:	11/11/2022
ISSUED:	DRAFT 2	DATE:	12/22/2022
ISSUED:	FINAL	DATE:	3/24/2023
ISSUED:		DATE:	

Engineer's Opinion of Probable Capital Cost
Municipal Biosolids - Pyrolysis/Gasification

Total Treatment Flow Rate			Units
Low	Middle	High	
2	6	10	dry US ton/day
4	9	15	MGD

			LOW FLOW RATE			MIDDLE FLOW RATE			HIGH FLOW RATE			
Cat. No.	**ITEM DESCRIPTION**	**UNIT**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**NOTES**
A	General Conditions	LS	1	$3,213,000	$3,213,000	1	$5,146,000	$5,146,000	1	$7,894,000	$7,894,000	1,2,3,4
B	PFAS Destruction Equipment	LS	1	$7,552,000	$7,552,000	1	$12,429,000	$12,429,000	1	$18,560,000	$18,560,000	1,2,3,4
C	Piping and Appurtenances	LS	1	$3,776,000	$3,776,000	1	$6,215,000	$6,215,000	1	$9,280,000	$9,280,000	1,2,3,4
D	Electrical and Instrumentation/Controls	LS	1	$1,511,000	$1,511,000	1	$2,486,000	$2,486,000	1	$3,712,000	$3,712,000	1,2,3,4
E	Treatment Building	SF	5,300	$500	$2,650,000	7,000	$500	$3,500,000	13,000	$500	$6,500,000	1,2,3,4
F	Site Work	LS	1	$265,000	$265,000	1	$350,000	$350,000	1	$650,000	$650,000	1,2,3,4
G	Installation (Equipment and Piping)	LS	1	$5,664,000	$5,664,000	1	$9,322,000	$9,322,000	1	$13,920,000	$13,920,000	1,2,3,4
	CONSTRUCTION SUBTOTAL				$24,630,000			$39,450,000			$60,520,000	1,2,3,4,5
	CONSTRUCTION CONTINGENCY			25%	$6,160,000		25%	$9,860,000		25%	$15,130,000	1,4,5
	ESTIMATED CONSTRUCTION COST				$30,790,000			$49,310,000			$75,650,000	1,2,3,4,5
	ESTIMATED TOTAL PROJECT COST				**$30,800,000**			**$49,400,000**			**$75,700,000**	1,2,3,4,6
	ESTIMATED ACCURACY RANGE	-30%			$21,600,000			$34,600,000			$53,000,000	4,6
		50%			$46,200,000			$74,100,000			$113,600,000	4,6

Notes

[1] Limited design work completed.

[2] Quantities based on design work completed.

[3] Unit prices based on information available at this time.

[4] This feasibility-level (Class 5, 0-2% design completion per AACE International Recommended Practice No. 17R-97) cost estimate is based on preliminary designs, quantities and unit prices. Costs will change with further design. Time value-of-money escalation costs are not included. Contingency is an allowance for the net sum of costs that will be in the Final Total Project Cost at the time of the completion of design, but are not included at this level of project definition. The estimated accuracy range for the Total Project Cost as the project is defined is -30% to +50%. The accuracy range is based on professional judgement considering the level of design completed, the complexity of the project and the uncertainties in the project as scoped. The contingency and the accuracy range are not intended to include costs for future scope changes that are not part of the project as currently scoped or costs for risk contingency. Operation and maintenance costs are not included.

[5] Estimate costs are reported to nearest ten thousand dollars.

[6] Estimate costs are rounded up to nearest one hundred thousand dollars. Estimate costs are to construct each alternative. The estimated costs do not include planning, engineering and design, permitting, construction administration, maintenance, monitoring or additional tasks following construction.

PREPARED BY: BARR ENGINEERING COMPANY

ENGINEER'S OPINION OF PROBABLE PROJECT COST

PROJECT: Evaluation of Current Alternatives and Estimated Costs for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water
LOCATION: MINNESOTA
PROJECT #: 23621432
OPINION OF COST - SUMMARY

SHEET:	1	OF	1
BY:	HAZEN	DATE:	12/9/2022
CHECKED BY:	ALL	DATE:	12/20/2022
APPROVED BY:	HAZEN	DATE:	3/15/2023
ISSUED:	DRAFT 1	DATE:	NA
ISSUED:	DRAFT 2	DATE:	12/22/2022
ISSUED:	FINAL	DATE:	3/24/2023
ISSUED:		DATE:	

Engineer's Opinion of Probable O&M Cost

Municipal Biosolids - Pyrolysis/Gasification

Total Treatment Flow Rate			Units
Low	Middle	High	
2.3	5.8	10.0	US dry ton/day

Item Num.	Description of O&M Category	Unit of Measure	Unit Rate (USD/unit)	LOW FLOWRATE Annual Quantity	LOW FLOWRATE Annual Total (USD)	MIDDLE FLOWRATE Annual Quantity	MIDDLE FLOWRATE Annual Total (USD)	HIGH FLOWRATE Annual Quantity	HIGH FLOWRATE Annual Total (USD)	Notes
1	**VARIABLE COSTS**									
1.1	**UTILITIES**				**$ 123,000**		**$ 220,000**		**$ 386,000**	1
1.1.1	Electricity - process equipment	kWh	$ 0.13	252,300	$ 34,000	636,200	$ 85,000	1,097,000	$ 147,000	1,2
1.1.2	Electricity - Non-process energy load (building power, HVAC, etc.)	kWh	$ 0.13	259,600	$ 35,000	342,800	$ 46,000	636,600	$ 85,000	1
1.1.3	Natural gas - Non-process energy load (heating)	MMBTU	$ 15.33	190	$ 3,000	250	$ 4,000	460	$ 8,000	1,2
1.1.4	Natural gas - Process fuel	MMBTU	$ 15.33	3,300	$ 51,000	5,500	$ 85,000	9,500	$ 146,000	1
1.2	**RESIDUALS DISPOSAL**				**$ 1,000**		**$ 2,000**		**$ 4,000**	1
1.2.1	Biochar management and disposal	tons	$ 100	10	$ 1,000	20	$ 2,000	40	$ 4,000	1
1.3	**MAINTENANCE**				**$ 38,000**		**$ 81,000**		**$ 144,000**	1
1.3.1	Process equipment maintenance, including materials	dry tons	$ 30	800	$ 24,000	2,100	$ 63,000	3,700	$ 111,000	1
1.3.2	General building maintenance	sq foot	$ 2.50	5,300	$ 14,000	7,000	$ 18,000	13,000	$ 33,000	1
1.4	**ANALYTICAL MONITORING**				**$ 12,000**		**$ 12,000**		**$ 12,000**	1
1.4.1	Monitoring - Monthly samples	SAMPLE	$ 400	30	$ 12,000	30	$ 12,000	30	$ 12,000	1
Item Num.	**Description of O&M Category**	**Unit of Measure**	**Annual Salary (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Notes**
2	**FIXED COSTS**									
2.1	**O&M LABOR**									
2.1.1	Water Treatment Operator	FTE	$ 100,000	0.25	$ 25,000	0.75	$ 75,000	1	$ 100,000	1
2.1.2	Shift Maintenance (Mechanical, Electrical)	FTE	$ 100,000	0.25	$ 25,000	0.5	$ 50,000	0.5	$ 50,000	1
	Total Workforce	**Base Payroll**			$ 50,000		$ 125,000		$ 150,000	1
		Fringe Rate	0.5		$ 25,000		$ 62,500		$ 75,000	1
		Total Payroll			**$ 75,000**		**$ 188,000**		**$ 225,000**	1
	ESTIMATED TOTAL ANNUAL OPERATING COSTS				**$ 250,000**		**$ 510,000**		**$ 780,000**	3
	Estimated Uncertainty Range	-30%			$ 180,000		$ 360,000		$ 550,000	3,4
		+50%			$ 380,000		$ 770,000		$ 1,170,000	3,4
	Unit Cost per dry ton treated				*$ 298*		*$ 241*		*$ 214*	

Notes

[1] Annual total cost values are rounded up to the nearest thousand USD.

[2] Assumed operating 7 days of operation per week, 365 days per year. Adjusted based on vendor-provided electricity consumption.

[3] Values less than $1 million are rounded to the nearest ten thousand USD, and values greater than or equal to $1 million are rounded up to the nearest one hundred thousand USD.

[4] Operation and maintenance costs are based on a Class 5 capital cost estimate with a +50/-30% uncertainty as applicable for projects at less than 2% of full project definition per AACE International 17R-97. Operation and maintenance costs are also expected to have a +50/-30% uncertainty.

Appendix E

SCWO – Municipal Wastewater Biosolids

Appendix E: Table 2 Solids	**Alternative Solids - SCWO**			**Units**	**Notes**
Waste stream	Solids	Solids	Solids	--	
Design flow rate	2	8	15	MGD	Biosolids industry standard: 1 mgd of influent flow produces 6 -7 dry ton/day of digested sludge.
Wet solids production rate	6	30	60	wet metric tons/day	15% total solids.
Dry solids production rate	1	5	10	dry US tons/day	
SCWO unit capacity	1	5	5	dry US tons/day	
Number of units	1	1	2	--	
Lab and bathroom	Yes	Yes	Yes	--	
Estimated footprint, total	1,500	2,200	3,800	ft2	Rounded to the nearest 100 ft2.

PREPARED BY: HAZEN on behalf of
BARR ENGINEERING COMPANY

ENGINEER'S OPINION OF PROBABLE PROJECT COST

PROJECT: Evaluation of Current Alternatives and Estimated Costs for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water
LOCATION: MINNESOTA
PROJECT #: 23621432
OPINION OF COST - SUMMARY

SHEET:	1	OF	
BY:	HAZEN	DATE:	10/10/2022
CHECKED BY:	ALL	DATE:	11/6/2022
APPROVED BY:	HAZEN	DATE:	3/15/2023
ISSUED:	DRAFT 1	DATE:	11/11/2022
ISSUED:	DRAFT 2	DATE:	12/22/2022
ISSUED:	FINAL	DATE:	3/24/2023
ISSUED:		DATE:	

Engineer's Opinion of Probable Capital Cost
Municipal Biosolids - SCWO

Total Treatment Flow Rate			Units
Low	Mid	High	
1	5	10	Dry US ton/day
2	8	15	MGD

			LOW FLOW RATE			MIDDLE FLOW RATE			HIGH FLOW RATE			
Cat. No.	**ITEM DESCRIPTION**	**UNIT**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**QUANTITY**	**UNIT COST**	**ITEM COST**	**NOTES**
A	General Conditions	LS	1	$1,213,000	$1,213,000	1	$2,955,000	$2,955,000	1	$5,909,000	$5,909,000	1,2,3,4
B	PFAS Destruction Equipment	LS	1	$3,188,000	$3,188,000	1	$7,668,000	$7,668,000	1	$15,335,000	$15,335,000	1,2,3,4
C	Piping and Appurtenances	LS	1	$1,594,000	$1,594,000	1	$3,834,000	$3,834,000	1	$7,668,000	$7,668,000	1,2,3,4
D	Electrical and Instrumentation/Controls	LS	1	$638,000	$638,000	1	$1,534,000	$1,534,000	1	$3,067,000	$3,067,000	1,2,3,4
E	Treatment Building	SF	500	$500	$250,000	1,650	$500	$825,000	3,300	$500	$1,650,000	1,2,3,4
F	Site Work	LS	1	$25,000	$25,000	1	$83,000	$83,000	1	$165,000	$165,000	1,2,3,4
G	Installation (Equipment and Piping)	LS	1	$2,391,000	$2,391,000	1	$5,751,000	$5,751,000	1	$11,502,000	$11,502,000	1,2,3,4
	CONSTRUCTION SUBTOTAL				$9,300,000			$22,650,000			$45,300,000	1,2,3,4,5
	CONSTRUCTION CONTINGENCY			25%	$2,330,000		25%	$5,660,000		25%	$11,330,000	1,4,5
	ESTIMATED CONSTRUCTION COST				$11,630,000			$28,310,000			$56,630,000	1,2,3,4,5
	ESTIMATED TOTAL PROJECT COST				**$11,700,000**			**$28,400,000**			**$56,700,000**	1,2,3,4,6
	ESTIMATED ACCURACY RANGE	-30%			$8,200,000			$19,900,000			$39,700,000	4,6
		50%			$17,600,000			$42,600,000			$85,100,000	4,6

Notes

[1] Limited design work completed.

[2] Quantities based on design work completed.

[3] Unit prices based on information available at this time.

[4] This feasibility-level (Class 5, 0-2% design completion per AACE International Recommended Practice No. 17R-97) cost estimate is based on preliminary designs, quantities and unit prices. Costs will change with further design. Time value-of-money escalation costs are not included. Contingency is an allowance for the net sum of costs that will be in the Final Total Project Cost at the time of the completion of design, but are not included at this level of project definition. The estimated accuracy range for the Total Project Cost as the project is defined is -30% to +50%. The accuracy range is based on professional judgement considering the level of design completed, the complexity of the project and the uncertainties in the project as scoped. The contingency and the accuracy

[5] Estimate costs are reported to nearest ten thousand dollars.

[6] Estimate costs are rounded up to nearest one hundred thousand dollars. Estimate costs are to construct each alternative. The estimated costs do not include planning, engineering and design, permitting, construction administration, maintenance, monitoring or

PREPARED BY: BARR ENGINEERING COMPANY		SHEET: 1	OF	1	
		BY: HAZEN		DATE:	12/9/2022
ENGINEER'S OPINION OF PROBABLE PROJECT COST		CHECKED BY: ALL		DATE:	12/14/2022
		APPROVED BY: HAZEN		DATE:	3/15/2023
PROJECT:	Evaluation of Current Alternatives and Estimated Costs for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water	ISSUED:	DRAFT 1	DATE:	NA
LOCATION:	MINNESOTA	ISSUED:	DRAFT 2	DATE:	12/22/2022
PROJECT #:	23621432	ISSUED:	FINAL	DATE:	3/24/2023
OPINION OF COST - SUMMARY		ISSUED:		DATE:	

Engineer's Opinion of Probable O&M Cost

Municipal Biosolids - SCWO

Total Treatment Flow Rate			Units
Low	Middle	High	
1	5	10	US dry ton/day

				LOW FLOWRATE		MIDDLE FLOWRATE		HIGH FLOWRATE		
Item Num.	**Description of O&M Category**	**Unit of Measure**	**Unit Rate (USD/unit)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Notes**
1	**VARIABLE COSTS**									
1.1	**UTILITIES**				**$ 22,000**		**$ 5,000**		**$ 10,000**	1
1.1.1	Electricity - Process equipment	kWh	$ 0.13	109,500	$ 15,000	-109,500	$ (15,000)	-219,000	$ (30,000)	1,2
1.1.2	Electricity - Non-process energy load (building power, HVAC, etc.)	kWh	$ 0.13	24,500	$ 4,000	80,800	$ 11,000	161,600	$ 22,000	1
1.1.3	Natural gas - Non-process energy load (heating)	MMBTU	$ 15.33	20	$ 1,000	60	$ 1,000	120	$ 2,000	1,2
1.1.4	Natural gas - Start-up fuel	MBTU/hr	$ 15.33	100	$ 2,000	520	$ 8,000	1,040	$ 16,000	1
1.2	**RESIDUALS DISPOSAL**				**$ -**		**$ 2,000**		**$ 4,000**	1
1.2.1	Ash management and disposal	tons	$ 100	0	$ -	20	$ 2,000	40	$ 4,000	1
1.3	**MISC Consumables**				**$ 500**		**$ 2,900**		**$ 5,800**	1
1.3.1	Chemical feed	tons	$ 145	4	$ 500	20	$ 2,900	40	$ 5,800	1
1.4	**MAINTENANCE**				**$ 98,000**		**$ 236,000**		**$ 470,000**	1
1.4.1	Process equipment maintenance, including materials	% equipment	3%	1	$ 96,000	1	$ 231,000	1	$ 461,000	1
1.4.2	General building maintenance	sq foot	$ 2.50	500	$ 2,000	1,700	$ 5,000	3,300	$ 9,000	1
1.5	**ANALYTICAL MONITORING**				**$ 12,000**		**$ 12,000**		**$ 12,000**	1
1.5.1	Monitoring - Monthly samples	SAMPLE	$ 400	30	$ 12,000	30	$ 12,000	30	$ 12,000	1
Item Num.	**Description of O&M Category**	**Unit of Measure**	**Annual Salary (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Annual Quantity**	**Annual Total (USD)**	**Notes**
2	**FIXED COSTS**									
2.1	**O&M LABOR**									
2.1.1	Water Treatment Operator	FTE	$ 100,000	0.25	$ 25,000	0.25	$ 25,000	0.5	$ 50,000	1
2.1.2	Shift Maintenance (Mechanical, Electrical)	FTE	$ 100,000	0.25	$ 25,000	0.25	$ 25,000	0.5	$ 50,000	1
	Total Workforce	**Base Payroll**			$ 50,000		$ 50,000		$ 100,000	1
		Fringe Rate	0.5		$ 25,000		$ 25,000		$ 50,000	1
		Total Payroll			**$ 75,000**		**$ 75,000**		**$ 150,000**	1
	ESTIMATED TOTAL ANNUAL OPERATING COSTS				**$ 210,000**		**$ 340,000**		**$ 660,000**	3
	Estimated Uncertainty Range	-30%			$ 150,000		$ 240,000		$ 470,000	3,4
		+50%			$ 320,000		$ 510,000		$ 990,000	3,4
	Unit Cost per dry ton treated				*$ 575*		*$ 186*		*$ 181*	

Notes

[1] Annual total cost values are rounded up to the nearest thousand USD.

[2] Assumed operating 7 days of operation per week, 365 days per year. Adjusted based on vendor-provided electricity consumption and generation.

[3] Values less than $1 million are rounded to the nearest ten thousand USD, and values greater than or equal to $1 million are rounded up to the nearest one hundred thousand USD.

[4] Operation and maintenance costs are based on a Class 5 capital cost estimate with a +50/-30% uncertainty as applicable for projects at less than 2% of full project definition per AACE International 17R-97. Operation and maintenance costs are also expected to have a +50/-30% uncertainty.

Environmental Science & Technology

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Viewpoint

A New OECD Definition for Per- and Polyfluoroalkyl Substances

Zhanyun Wang,* Andreas M. Buser, Ian T. Cousins, Silvia Dematteo, Wiebke Drost, Olof Johansson, Koichi Ohno, Grace Patlewicz, Ann M. Richard, Glen W. Walker, Graham S. White, and Eeva Leinala

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KEYWORDS: *PFAS definition, cheminformatic tools, PFAS synthesis, PFAS universe, PFAS terminology*

Per- and polyfluoroalkyl substances (PFASs) comprise a class of chemicals that has attracted much attention since the early 2000s, when the hazards and ubiquitous occurrence of two PFASs—perfluorooctanoic acid (PFOA) and perfluorooctanesulfonic acid (PFOS)—were reported. Early communications used multiple terms such as "per- and polyfluorinated chemicals", "perfluorinated organics", "perfluorochemical surfactants", and "highly fluorinated compounds". In 2011, to harmonize communication, Buck et al.[1] published a milestone paper, providing the first clear structural definition of PFASs and recommendations on the names and acronyms for over 200 individual PFASs. Since then, research and regulation has expanded from PFOA and PFOS to a much wider range of substances.

In 2018, the so-called "Global PFC Group" led by the Organisation for Economic Co-operation and Development (OECD) and the United Nations Environment Programme (UNEP) published a list of over 4700 PFASs that contain a $-C_nF_{2n}-$ ($n \geq 3$) or $-C_nF_{2n}OC_mF_{2m}-$ (n and $m \geq 1$) moiety and that were known or likely to have been on the global market.[2] The list included substances that contain fully fluorinated carbon moieties, but do not meet the PFAS definition in Buck et al. (2011) due to a lack of a $-CF_3$ group in the molecule. Additionally, recent advancement of nontargeted analytical techniques enabled identification of many unknown PFASs in environmental and product samples. These developments provided motivation to reconcile the terminology of the PFAS universe, including a renewed look at the PFAS definition.

Against this backdrop, a report on the terminology of PFASs was recently published under the framework of the Global PFC Group.[3] This report reflects a three-year multistakeholder

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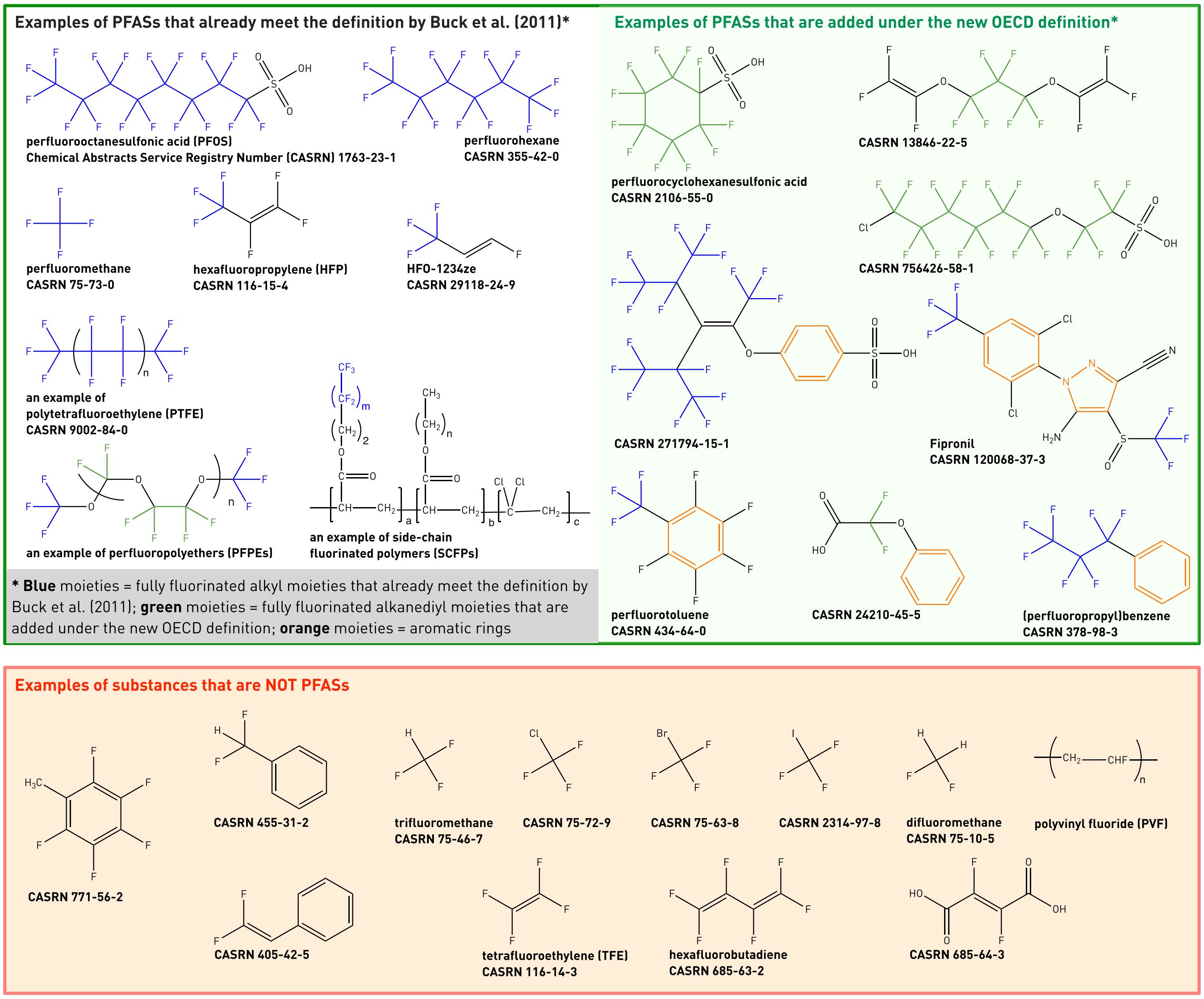

Figure 1. Examples of substances that are, and are not, PFASs based on the new definition.

international effort in reviewing the universe and terminology of PFASs to provide recommendations and practical guidance. We encourage stakeholders from academia, civil society, industry, and government to read the report and consider adopting its recommendations, wherever feasible, to help enable the coherent and consistent use of PFAS terminology across sectors and around the world. This Viewpoint provides an overview of the report.

■ A REVISED PFAS DEFINITION

The report details four major limitations with the previous definition in representing the PFAS universe: (1) omission of substances that have functional groups on both ends of the fully fluorinated carbon moiety (e.g., perfluoroalkyldicarboxylic acids); (2) inconsistencies in dealing with homologues that are fully fluorinated aliphatic cyclic compounds with or without a fully fluorinated alkyl side chain; (3) omission of substances with aromatic ring(s) in the nonfluorinated functional group(s) that can be cleaved in the environment and biota; and (4) use of the ambiguous term “highly fluorinated”.

To address these concerns, the report presents a revised, broadly inclusive PFAS definition: “*PFASs are defined as fluorinated substances that contain at least one* **fully fluorinated methyl or methylene carbon atom (without any H/Cl/Br/I atom attached to it)**, *i.e., with a few noted exceptions, any chemical with at least a perfluorinated methyl group* ($-CF_3$) *or a perfluorinated methylene group* ($-CF_2-$) *is a PFAS*”. The “noted exceptions” refer to a carbon atom with a H/Cl/Br/I atom attached to it.

The rationale behind the revision is to have a coherent and consistent definition across compounds from the view of the chemical structure. The new definition was required to be easily implementable for distinguishing between PFASs and non-PFASs, and easily understood by experts and nonexperts alike. Figure 1 illustrates examples that are, and are not, PFASs. This revised definition captures the broadness of the PFAS universe, ranging from small molecules, to more complex aromatics with a perfluorinated methyl/methylene group on the side chain(s), to diverse polymers.

Building on the revised definition, the report further provides (1) an explanation of how PFASs relate to other organofluorine compounds, (2) a comprehensive overview of known PFAS groups and their structural traits, including examples and notes on whether common names and acronyms exist, and (3) some common synthesis routes of individual or groups of PFASs.

PRACTICAL GUIDANCE ON HOW TO USE THE PFAS TERMINOLOGY

The report highlights the need to distinguish between the general definition and user-specific working scopes of PFASs. The general definition is based on molecular structure alone and serves as a starting and reference point to guide individual users to have a comprehensive understanding of the scale and diversity of chemicals in the PFAS universe. Meanwhile, users may define their own working scope of PFASs for specific activities according to their specific needs by combining the general definition with additional considerations (e.g., specific properties, use areas). When such a working scope of PFASs is used, the report recommends that respective users clearly provide the context and rationale for selecting their working scope to ensure transparency and avoid confusion by others.

The report further recommends using and building upon existing common terminologies such as in Buck et al. (2011) and well-defined common practices in organic chemistry, unless it is essential to deviate from them, to keep the consistent and coherent use of the PFAS terminology. As PFASs are a chemical class with diverse molecular structures and thus properties, it is recommended to properly recognize and communicate such diversity in a clear, specific and descriptive manner.

A SYSTEMATIC MOLECULAR STRUCTURE-BASED APPROACH TO CHARACTERIZING PFASS

When users define their own working scope of PFASs, they need to determine whether a compound falls or does not fall into their working scope. However, given the complexity and diversity of PFASs, it can be a challenging task to characterize and categorize PFASs based on chemical structures in a coherent and consistent manner, particularly for nonexperts. Different users may have different construction of working scopes, and there is no single categorization/grouping system that suits all. Therefore, the report provides a standardized approach for systematically characterizing PFASs based on molecular structural traits that will allow stakeholders to make their own categorization in a coherent and consistent manner. This system can be used to manually characterize and categorize PFASs, but the approach could also be used as inputs for developing automated cheminformatic tools.[4]

FUTURE WORK ON PFAS TERMINOLOGY

Four areas are recognized for further work to facilitate clear and unambiguous communication: (1) a centralized PFAS nomenclature database/platform; (2) development of chem-informatic tools for automated, structure-based systematic characterizaing and categorizing of PFASs; (3) work on the characterization and reporting of polymers; and (4) work on organofluorine compounds not currently defined as PFASs, including many fluorinated aromatics.

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Notes

The authors declare no competing financial interest.

Biography

Dr. Zhanyun Wang is a senior researcher in the Ecological Systems Design Group at ETH Zürich, Switzerland. As an environmental chemist by training, his research interests focus primarily on understanding the life cycles and risks of various anthropogenic chemicals in the technosphere and natural environment. He is also very interested in exploring novel and pragmatic approaches to advancing sound chemicals management, enabling a sustainable circular economy, and strengthening science-policy interface on chemicals and waste.

■ ACKNOWLEDGMENTS

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■ REFERENCES

(1) Buck, R. C.; Franklin, J.; Berger, U.; Conder, J. M.; Cousins, I. T.; de Voogt, P.; Jensen, A. A.; Kannan, K.; Mabury, S. A.; van Leeuwen, S. P. J. Perfluoroalkyl and polyfluoroalkyl substances in the environment: terminology, classification and origins. *Integr. Environ. Assess. Manage.* **2011**, *7* (4), 513−541.

(2) Organisation for Economic Co-operation and Development (OECD). 2018. Toward a new comprehensive global database of per- and polyfluoroalkyl substances (PFASs). http://www.oecd.org/chemicalsafety/risk-management/global-database-of-per-and-polyfluoroalkyl-substances.xlsx; http://www.oecd.org/officialdocuments/publicdisplaydocumentpdf/?cote=ENV-JM-MONO(2018)7&doclanguage=en (methodology report) (021/9/1).

(3) Organisation for Economic Co-operation and Development (OECD). 2021. Reconciling Terminology of the Universe of Per- and Polyfluoroalkyl Substances: Recommendations and Practical Guidance. Series on Risk Management No. 61. https://www.oecd.org/chemicalsafety/portal-perfluorinated-chemicals/terminology-per-and-polyfluoroalkyl-substances.pdf (accessed 2021/9/1)

(4) Sha, B.; Schymanski, E. L.; Ruttkies, C.; Cousins, I. T.; Wang, Z. Exploring open cheminformatics approaches for categorizing per- and polyfluoroalkyl substances (PFASs). *Environmental Science: Processes & Impacts* **2019**, *21* (11), 1835−1851.

Title 40 —Protection of Environment

Chapter I —Environmental Protection Agency

Subchapter R —Toxic Substances Control Act

Part 705 —Reporting and Recordkeeping Requirements for Certain Per- and Polyfluoroalkyl Substances

Authority: 15 U.S.C. 2607(a)(7).

Source: 88 FR 70548, Oct. 11, 2023, unless otherwise noted.

§ 705.3 Definitions.

The definitions in this section and the definitions in TSCA section 3 apply to this part. In addition, the definitions in 40 CFR 704.3 also apply to this part, except the definition for *small quantities solely for research and development*.

Article means a manufactured item which:

(1) Is formed to a specific shape or design during manufacture;

(2) Has end use function(s) depending in whole or in part upon its shape or design during end use; and

(3) Has either no change of chemical composition during its end use or only those changes of composition which have no commercial purpose separate from that of the article, and that result from a chemical reaction that occurs upon end use of other chemical substances, mixtures, or articles; except that fluids and particles are not considered articles regardless of shape or design.

Central Data Exchange or *CDX* means EPA's centralized electronic submission receiving system.

Chemical Information Submission System or *CISS* means EPA's electronic, web-based reporting tool for the completion and submission of data, reports, and other information, or its successors.

Commercial use means the use of a chemical substance or a mixture containing a chemical substance (including as part of an article) in a commercial enterprise providing saleable goods or services.

Consumer use means the use of a chemical substance or a mixture containing a chemical substance (including as part of an article) when sold to or made available to consumers for their use.

Environmental or health effects information means any information of any effect of a chemical substance or mixture containing a chemical substance on health or the environment or on both. This includes all health and safety studies.

(1) Not only is information that arises as a result of a formal, disciplined study included, but other information relating to the effects of a chemical substance or mixture containing a chemical substance on health or the environment is also included. Any information that bears on the effects of a chemical substance on health or the environment would be included.

(2) Examples are:

(i) Long- and short-term tests of mutagenicity, carcinogenicity, or teratogenicity; data on behavioral disorders; dermatoxicity; pharmacological effects; mammalian absorption, distribution, metabolism, and excretion; cumulative, additive, and synergistic effects; and acute, subchronic, and chronic effects.

(ii) Tests for ecological or other environmental effects on invertebrates, fish, or other animals, and plants, including acute toxicity tests, chronic toxicity tests, critical life-stage tests, behavioral tests, algal growth tests, seed germination tests, plant growth or damage tests, microbial function tests, bioconcentration or bioaccumulation tests, and model ecosystem (microcosm) studies.

(iii) Assessments of human and environmental exposure, including workplace exposure, and impacts of a particular chemical substance or mixture containing a chemical substance on the environment, including surveys, tests, and studies of: Biological, photochemical, and chemical degradation; structure/activity relationships; air, water, and soil transport; biomagnification and bioconcentration; and chemical and physical properties, *e.g.,* boiling point, vapor pressure, evaporation rates from soil and water, octanol/water partition coefficient, and water solubility.

(iv) Monitoring data, including but not limited to when they have been aggregated and analyzed to measure the exposure of humans or the environment to a chemical substance or mixture containing a chemical substance.

Health and safety studies means any study of any effect of a chemical substance or mixture on health or the environment or on both, including underlying information and epidemiological studies, studies of occupational exposure to a chemical substance or mixture, toxicological, clinical, and ecological studies of a chemicals substance or mixture containing a chemical substance, and any test performed under TSCA. The following information is not part of a health and safety study:

(1) The name, address, or other identifying information for the submitting company, including identification of the laboratory that conducted the study in cases where the laboratory is part of or closely affiliated with the submitting company;

(2) Internal product codes (*i.e.,* code names for the test substance used internally by the submitting company or to identify the test substance to the test laboratory);

(3) Names and contact details for testing laboratory personnel and names and other private information for health and safety study participants or persons involved in chemical incidents such as would typically be withheld under 5 U.S.C. 552(b)(6) or under other privacy laws; and

(4) Information pertaining to test substance product development, advertising, or marketing plans, or to cost and other financial data.

Highest-level U.S. parent company means the highest-level company of the site's ownership hierarchy as of the start of the submission period during which data are being reported according to the following instructions. The highest-level U.S. parent company is located within the United States. The following rules govern how to identify the highest-level U.S. parent company:

(1) If the site is entirely owned by a single U.S. company that is not owned by another company, that single company is the U.S. parent company.

(2) If the site is entirely owned by a single U.S. company that is, itself, owned by another U.S.-based company (*e.g.,* it is a division or subsidiary of a higher-level company), the highest-level domestic company in the ownership hierarchy is the U.S. parent company.

(3) If the site is owned by more than one company (*e.g.,* company A owns 40 percent, company B owns 35 percent, and company C owns 25 percent), the company with the largest ownership interest in the site is the U.S. parent company. If a higher-level company in the ownership hierarchy owns more than

one ownership company, then determine the entity with the largest ownership by considering the lower-level ownerships in combination (*e.g.,* corporation X owns companies B and C, for a total ownership of 60 percent for the site).

(4) If the site is owned by a 50:50 joint venture or a cooperative, the joint venture or cooperative is its own parent company. If the site is owned by a U.S. joint venture or cooperative, the highest level of the joint venture or cooperative is the U.S. parent company.

(5) If the site is federally owned, the highest-level Federal agency or department is the U.S. parent company.

(6) If the site is owned by a non-Federal public entity, that entity (such as a municipality, State, or tribe) is the U.S. parent company.

Industrial function means the intended physical or chemical characteristic for which a chemical substance or mixture is consumed as a reactant; incorporated into a formulation, mixture, reaction product or article; repackaged; or used.

Industrial use means use at a site at which one or more chemical substances or mixtures are manufactured (including imported) or processed.

Intended for use by children means the chemical substance or mixture is used in or on a product that is specifically intended for use by children aged 14 or younger. A chemical substance or mixture containing a chemical substance is intended for use by children when the submitter answers "yes" to at least one of the following questions for the product into which the submitter's chemical substance or mixture containing a chemical substance is incorporated:

(1) Is the product commonly recognized (*i.e.,* by a reasonable person) as being intended for children aged 14 or younger?

(2) Does the manufacturer of the product state through product labeling or other written materials that the product is intended for or will be used by children aged 14 or younger?

(3) Is the advertising, promotion, or marketing of the product aimed at children aged 14 or younger?

Known to or reasonably ascertainable by means all information in a person's possession or control, plus all information that a reasonable person similarly situated might be expected to possess, control, or know.

Manufacture means to import into the customs territory of the United States (as defined in general note 2 of the Harmonized Tariff Schedule of the United States (19 U.S.C. 1202)), produce, or manufacture for commercial purposes.

Manufacture for commercial purposes means:

(1) To import, produce, or manufacture with the purpose of obtaining an immediate or eventual commercial advantage for the manufacturer, and includes among other things, such "manufacture" of any amount of a chemical substance or mixture containing a chemical substance:

(i) For commercial distribution, including for test marketing; and/or

(ii) For use by the manufacturer, including use for product research and development, or as an intermediate.

(2) Manufacture for commercial purposes also applies to substances that are produced coincidentally during the manufacture, processing, use, or disposal of another substance or mixture containing a chemical substance, including both byproducts that are separated from that other substance or mixture containing a chemical substance and impurities that remain in that substance or mixture containing a chemical substance. Such byproducts and impurities may, or may not, in themselves have commercial value. They are nonetheless produced for the purpose of obtaining a commercial advantage since they are part of the manufacture of a chemical product for a commercial purpose.

Per- and polyfluoroalkyl substances or *PFAS* means, for the purpose of this part, any chemical substance or mixture containing a chemical substance that structurally contains at least one of the following three sub-structures:

(1) $R-(CF_2)-CF(R')R''$, where both the CF_2 and CF moieties are saturated carbons.

(2) $R-CF_2OCF_2-R'$, where R and R' can either be F, O, or saturated carbons.

(3) $CF_3C(CF_3)R'R''$, where R' and R" can either be F or saturated carbons.

Possession or control means in possession or control of the submitter, or of any subsidiary, partnership in which the submitter is a general partner, parent company, or any company or partnership which the parent company owns or controls, if the subsidiary, parent company, or other company or partnership is associated with the submitter in the research, development, test marketing, or commercial marketing of the chemical substance in question. (A parent company owns or controls another company if the parent owns or controls 50 percent or more of the other company's voting stock. A parent company owns or controls any partnership in which it is a general partner.) Information is included within this definition if it is:

(1) In files maintained by submitter's employees who are:

(i) Associated with research, development, test marketing, or commercial marketing of the chemical substance in question; and/or

(ii) Reasonably likely to have such data.

(2) Maintained in the files of other agents of the submitter who are associated with research, development, test marketing, or commercial marketing of the chemical substance in question in the course of their employment as such agents.

Research and development (R&D) means activities intended solely as scientific experimentation, research, or analysis. R&D focuses on the analysis of the chemical or physical characteristics, the performance, or the production characteristics of a chemical substance, a mixture containing the substance, or an article. R&D encompasses a wide range of activities which may occur in a laboratory, pilot plant, commercial plant outside the research facility, or at other sites appropriate for R&D. General distribution of chemical substances to consumers does not constitute R&D.

Site-limited means a chemical substance is manufactured and processed only within a site and is not distributed as a chemical substance or as part of a mixture or article containing a chemical substance outside the site. Imported chemical substances are never site-limited.

Worker means someone at a site of manufacture, import, or processing who performs work activities near sources of a chemical substance or mixture or directly handles the chemical substance or mixture during the performance of work activities.

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ORIGINAL PAPER

Leveraging Systematic Reviews to Explore Disease Burden and Costs of Per- and Polyfluoroalkyl Substance Exposures in the United States

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Abstract

Accelerating evidence confirms the contribution of per- and polyfluoroalkyl substances (PFAS) to disease burden and disability across the lifespan. Given that policy makers raise the high cost of remediation and of substituting PFAS with safer alternatives in consumer products as barriers to confronting adverse health outcomes associated with PFAS exposure, it is important to document the costs of inaction even in the presence of uncertainty. We therefore quantified disease burdens and related economic costs due to legacy PFAS exposures in the US in 2018. We leveraged systematic reviews and used meta-analytic inputs whenever possible, identified previously published exposure–response relationships, and calculated PFOA- and PFOS-attributable increases in 13 conditions. These increments were then applied to census data to determine total annual PFOA- and PFOS-attributable cases of disease, from which we calculated economic costs due to medical care and lost productivity using previously published cost-of-illness data. We identified PFAS-attributable disease costs in the US of \$5.52 billion across five primary disease endpoints shown to be associated with PFAS exposure in meta-analyses. This estimate represented the lower bound, with sensitivity analyses revealing as much as \$62.6 billion in overall costs. While further work is needed to assess probability of causation and establish with greater certainty effects of the broader category of PFAS, the results confirm further that public health and policy interventions are still necessary to reduce exposure to PFOA and PFOS and their endocrine-disrupting effects. This study demonstrates the large potential economic implications of regulatory inaction.

Keywords PFAS · Perfluoroalkyl substances · Polyfluoroalkyl substances · Environmental chemicals · Disease burden · Economic costs · Obesity · Diabetes · Metabolism · Cancer · Reproductive health · Fertility · Respiratory infection · Child health

Abbreviations

AF	Attributable fraction
ART	Assisted reproductive technology
BMI	Body mass index
CI	Confidence interval
DALY	Disability-adjusted life years
ERR	Exposure–response relationship
GDM	Gestational diabetes
IQ	Intelligence quotient
LBW	Low birth weight
NHANES	National Health and Nutrition Examination Survey
OR	Odds ratio
PCOS	Polycystic ovarian syndrome
PFAS	Perfluoroalkyl substances
PFOA	Perfluorooctanoic acid
PFOS	Perfluorooctane sulfonic acid
ppt	Parts per trillion
QALY	Quality-adjusted life years
RR	Risk ratio
SD	Standard deviation
TTP	Time to pregnancy
T2D	Type 2 diabetes mellitus
US	United States

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Introduction

Per- and polyfluoroalkyl substances (PFAS) are a group of over 4700 human-made fluorine-rich molecules (Birnbaum 2018). Long-chain PFAS, with a minimum of six carbons in their "backbone," were first developed in the 1940s. The polarity of their structure enhanced their utility in the production of water- and oil-resistant clothing, electronics, nonstick cookware, carpets, and food packaging materials for many years (Arbuckle et al. 2013; Holzer et al. 2008). These chemicals are widely detected in the blood of human populations worldwide (Bach et al. 2016a; Calafat et al. 2007), in part due to the biological persistence of many long-chain PFAS, such as perfluorooctanoic acid (PFOA) and perfluorooctane sulfonic acid (PFOS), which have half-lives in humans of at least 2 years (Bartell et al. 2010; Olsen et al. 2007; Xu et al. 2020). Although PFOA and PFOS have been added to the Stockholm Convention and PFOA use has been banned in the EU, they are still being released into the environment and are still being produced in other countries. Furthermore, both chemicals persist in the environment due to their chemical stability, resulting in ongoing human exposure (Grandjean and Clapp 2015).

Among the first to document PFAS-related effects on human health were the C8 Science Panel exposure and health studies conducted between 2005 and 2013 in mid-Ohio Valley communities where PFOA had heavily contaminated the water supply since the 1950s. These studies identified probable links with diagnosed high cholesterol, ulcerative colitis, thyroid disease, testicular cancer, kidney cancer, and pregnancy-induced hypertension (C8 Science Panel). An updated report from C8 Science Panel members and colleagues suggests that while the epidemiologic evidence for some of the associations they identified remains limited, possibly due to lower exposure levels in the general population, their findings for high cholesterol, ulcerative colitis, and kidney and testicular cancer had been reinforced by subsequent studies and impaired immune function had emerged as an additional outcome (Steenland et al. 2020). A recent scoping review of studies exclusively conducted among general population samples concluded that the weight of evidence supported associations of low-level PFAS exposure with low birth weight (LBW, < 2500 g), childhood obesity, adult obesity, adult-onset type 2 diabetes (T2D), gestational diabetes (GDM), endometriosis, polycystic ovarian syndrome (PCOS), couple infertility, and breast cancer (Kahn et al. 2020). Systematic reviews add further support for routine PFAS exposure and LBW (Bach et al. 2015; Johnson et al. 2014; Koustas et al. 2014; Lam et al. 2014; Steenland et al. 2018); childhood obesity (Liu et al. 2018b), dyslipidemia (Rappazzo et al. 2017), renal dysfunction (Rappazzo et al. 2017), respiratory infection (Rappazzo et al. 2017), and reduced immune response to vaccines (Rappazzo et al. 2017); age at menarche (Rappazzo et al. 2017); and adult thyroid dysfunction (Kim et al. 2018) and kidney (Bartell and Vieira 2021), testicular (Bartell and Vieira 2021), and breast (Wan et al. 2021) cancers. While a 2016 systematic review cast doubt on evidence for infertility due to PFAS because most of the studies that found associations were not restricted to nulliparous women (Bach et al. 2016b), the authors acknowledged that four of eight studies identified increased time to pregnancy (TTP) with PFOA or PFOS exposure. A more recent scoping review was less dismissive of the evidence and pointed out that studies conducted among parous women may still be valid if models adjust for interpregnancy interval and (in retrospective studies) gestational age at blood collection (Kahn et al. 2021). A difference-in-difference analysis of a natural experiment in which fertility and birth outcomes were compared between communities without PFAS exposure and highly exposed communities where PFOA- and PFOS-contaminated water supplies were remediated found that preterm birth and LBW rates, which had been higher in the contaminated communities, decreased following remediation and the fertility rate, which had been lower, increased (Waterfield et al. 2020).

In the United States (US), rising concerns about the health effects of PFAS have prompted calls to state and federal governments to limit ongoing PFAS use and remediate contaminated water supplies. The US Environmental Protection Agency's third Unregulated Containment Monitoring Rule report released in January 2017 found that 4% of water systems reported at least one PFAS compound detectable above the minimum reporting level, which ranged from 10 to 90 parts per trillion (ppt) for various PFAS (Crone et al. 2019). A more recent study estimates that 18–80 million people in the US receive tap water with at least 10 ppt of PFOA and PFOS combined and more than 200 million Americans have tap water contaminated with PFOA and PFOS concentrations of 1 ppt or higher (Andrews and Naidenko 2020). Although there is currently no national regulatory limit for PFOA and PFOS exposure and the US Environmental Protection Agency continues to use a lifetime health advisory level of 70 ppt for the sum of PFOA and PFOS (Environmental Protection Agency 2016), some states have banned PFAS in food packaging and lowered regulatory limits for PFAS in drinking water by two orders of magnitude to 1 ppt or lower as suggested by studies of PFAS and antibody titers in children (Grandjean and Clapp 2015; Hoylman 2020).

In considering regulatory action, the European Food Safety Authority and the US Agency for Toxic Substances and Disease Registry have suggested that the evidence is not sufficient to confirm causality and therefore to proceed with steps to reduce exposure (Rogers et al. 2021; Schrenk et al. 2020). As Bradford Hill declared in his landmark lecture on causal inference (Hill 1965), uncertainty "does not confer upon us a freedom to ignore the knowledge we already have or to postpone the action that it appears to demand at a given time." Given that policy makers raise the high cost of remediation and of substituting PFAS with safer alternatives in consumer products as barriers to confronting adverse health outcomes associated with PFAS exposure, it is important to document the costs of inaction even in the presence of uncertainty.

Recent studies suggest that the disease-related burden due to PFAS can be substantial. In 2003–2004, PFOA exposure accounted for up to 4% of LBW in the US, with $13.7 billion in associated costs (Malits et al. 2018). The Nordic Council of Ministers estimated €52–84 billion in disease-related costs in 2019 associated with PFAS within the European Economic Area, driven substantially by occupational PFAS exposures and effects on populations living near contaminated sites (Goldenman et al. 2019). Yet, these analyses did not consider the broader scope of health effects of routine, low-level environmental PFAS exposure on human health. The aim for this analysis was therefore to quantify disease burden and associated costs of PFOA and PFOS exposure among the entire US population based on health outcomes with a substantial weight of evidence in support of associations with PFAS exposure.

Methods

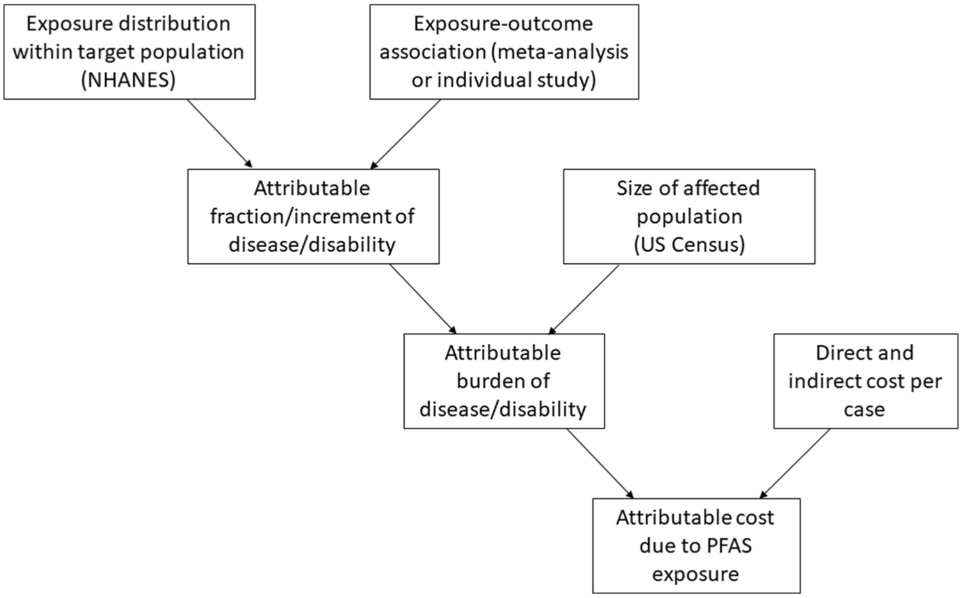
Overall approach (Fig. 1)

To identify diseases and dysfunctions and their associated economic costs to be considered for possible attribution to PFAS exposure, we leveraged the PFAS-Tox Database (https://pfastoxdatabase.org/), which was built using systematic review methods, to extract relevant studies and data (Pelch et al. 2019). Given the rapidly evolving nature of the PFAS literature, we supplemented the primary source with a PubMed search using the terms "PFAS" and "systematic review" or "meta-analysis." In an additional effort to be complete, we also mined three recent scoping reviews to ensure the most comprehensive inclusion of potential disease burden and costs in sensitivity analyses (Kahn et al. 2020; Steenland et al. 2020; Steenland and Winquist 2021).

In main estimates of PFAS-attributable disease burden and cost, we only considered disease outcomes for which statistically significant associations had been derived from published meta-analyses of epidemiologic studies. These included (1) LBW due to prenatal exposure (Steenland et al. 2018); (2) childhood obesity due to prenatal exposure (Liu et al. 2018b); (3) kidney cancer due to lifetime exposure (Bartell and Vieira 2021); (4) testicular cancer due to lifetime exposure (Bartell and Vieira 2021); and (5) hypothyroidism in females due to lifetime exposure (Kim et al. 2018). For all of these outcomes except hypothyroidism, we used the meta-analytic estimates of exposure–response relationship [ERR, e.g., odd ratios (ORs) or risk ratios (RRs)] as the bases for disease burden and cost estimations. For hypothyroidism, a clear negative association was demonstrated between PFOA exposure and total T4 and T3 levels in a meta-analysis of seven papers by Kim et al., but because these are not clinical endpoints, we used an OR from a representative paper from the meta-analysis that identified a negative association with subclinical hypothyroidism (Wen et al. 2013).

In a sensitivity analysis of PFAS-attributable disease burden and cost, we expanded the scope of relevant outcomes

Fig. 1 Schematic of method for calculating cost of disease/disability attributable to PFAS exposure



to consider health conditions for which relations with PFAS had been identified in systematic and scoping reviews but had not been meta-analyzed. These included (6) adult obesity due to exposure over the lifespan (Kahn et al. 2020); (7) T2D in females due to exposure over the lifespan (Kahn et al. 2020); (8) GDM due to exposure measured in pregnancy (Kahn et al. 2020); (9) endometriosis due to exposure over the lifespan (Kahn et al. 2020); (10) PCOS due to exposure over the lifespan (Kahn et al. 2020); (11) couple infertility due to lifetime exposure in females (Kahn et al. 2020); (12) female breast cancer due to lifetime exposure (Wan et al. 2021); and (13) pneumonia in children due to prenatal exposure (Rappazzo et al. 2017). We did not include pediatric dyslipidemia or reduced age at menarche (Rappazzo et al. 2017), as these indicators are associated with outcomes already included in our analysis (e.g., childhood obesity and breast cancer, respectively); similarly, we did not include adult high cholesterol (Steenland et al. 2020), as it is associated with adult obesity, which is already included in our analysis. We also did not include reduced response to childhood vaccination (Grandjean et al. 2012), as reduced titers generally only require revaccination and clinical episodes of tetanus [~ 30 cases per year (CDC)] and diphtheria [2 cases between 2004 and 2017 (CDC)] are extremely rare in the US (CDC 2022a, b). Recognizing that some studies for each of the included outcomes might have reported null findings, the lower bound of economic cost added for this group of outcomes is zero. We based the upper bound of the sensitivity analysis on ERRs drawn from recent well-designed studies that reported statistically significant results from populations most similar to the current US population and extracted appropriate ERRs for our exposures and outcomes of interest (Tables 1, 2). To extrapolate most accurately effects in 2018 [the most recent year for which PFAS exposure data are available from the US National Health and Nutrition Examination Survey (NHANES)], we considered only studies published within the past 10 years and excluded those that did not control for confounding variables in the analysis, did not have PFAS exposure levels similar to our population as defined by the 2017–2018 NHANES dataset, and did not provide an RR, OR, or beta coefficient with either a 95% confidence interval or *p*-value. When multiple studies met these criteria, we modeled each separately and added the highest estimate to our cost estimate total in order to establish the upper bound of our sensitivity analysis.

Assessing Risk of Bias

A substantial literature has described and compared methods to evaluate systematic reviews (Whiting et al. 2016) and epidemiologic studies (Eick et al. 2020) for risk of bias. We used the tool developed by the National Toxicology Program's Office of Health Assessment and Translation (OHAT) (Office of Health Assessment and Translation (OHAT) 2022) to evaluate epidemiologic studies and ROBIS, the first rigorously developed tool designed specifically to assess the risk of bias in systematic reviews (Whiting et al. 2016). Two authors (LT, LK) independently evaluated each of the studies.

OHAT includes seven questions that yield graded probability assessments for risk of bias within observational studies (definitely low, probably low, probably high, definitely high). In the cases where there was a potential risk of bias, we added narrative comments to explain reasons for our concerns. ROBIS evaluates risk of bias in systematic reviews across four domains: study eligibility criteria; identification and selection of studies; data collection and study appraisal; and synthesis and findings. Within each domain, answers to multiple questions are used to assemble a domain-wide assessment of risk of bias (low, high, unclear). For each systematic review, two authors (LT and LK) assessed overall risk of bias in each of the four domains, identified specific concerns, and then assessed whether conclusions were supported by the evidence based on three criteria: whether the interpretation of findings addressed identified concerns in all the domains; whether the relevance of identified studies to the research question was appropriately considered; and whether authors overemphasized statistical significance. These questions were answered as yes, probably yes, probably no, no, or no information. This informed final assessments of each systematic review as low, high, or unclear.

Estimating PFAS-Attributable Disease Burden and Cost

To estimate the attributable cost of PFAS-mediated disease, we applied the model first used by the Institute of Medicine (1981) described by the equations below:

$$\text{Attributable disease burden} = \text{Increment in disease/disability} \times \text{Attributable fraction(AF)} \times \text{Population size} \tag{1}$$

$$\text{Attributable cost} = \text{Attributable disease burden} \times \text{Cost per increment.} \tag{2}$$

Table 1 Study selection

Exposure	Author (Year)	N	Study type	Date of recruitment	Location	Population	Exposure	Outcome	Covariates	Results
Low birth weight										
PFOA~	Steenland (2018)	24 studies	Meta-analysis	Varied by study	Varied by study	Varied by study	PFOA in maternal or cord blood	Birth weight	Varied by study	10.5 g (4.4, 16.7) decrease in birth weight per ng/mL PFOA increase in maternal or cord blood
PFOS*	Meng (2018)	3535 mother–infant pairs	Cross-sectional	1992–2002	Denmark	Mother–infant pairs	PFOS in maternal plasma	Birth weight	Infant sex, infant birth year, gestational week of blood draw, maternal age, parity, socio-occupational status, pre-pregnancy BMI, smoking, and alcohol use during pregnancy	45.2 g (13.6, 76.8) decrease in birth weight per doubling of ng/mL PFOS increase in maternal plasma
Childhood obesity at age 10										
PFOA~	Liu (2018a)	Nine studies with prenatal exposure	Meta-analysis	Varied by study	Varied by study	Multiple cohorts of children	PFOA in early childhood	BMI	Varied by study	0.09 (0.02, 0.17) increase in BMI *z*-score per ng/mL increase in prenatal PFOA
PFOS*	Lauritzen (2018)	412 females	Prospective cohort	1986–1988	Norway and Sweden	Pregnant women	PFOS in maternal serum	BMI	Maternal age, education, smoking at conception, pre-pregnancy BMI, weight gain at 17 weeks, interpregnancy interval, previous breastfeeding duration, and country of residence	0.18 (0.01, 0.35) increase in BMI *z*-score per ln-unit ng/mL increase in prenatal PFOA

Table 1 (continued)

Exposure	Author (Year)	N	Study type	Date of recruitment	Location	Population	Exposure	Outcome	Covariates	Results
Kidney cancer										
PFOA~	Bartell (2021)	Four studies	Meta-analysis	Varied by study	Varied by study	Varied by study	PFOA exposure	Kidney cancer incidence	Varied by study	Increase in cancer risk per 10 ng/mL increase in serum PFOA = 16% (3%, 30%)
Testicular cancer										
PFOA~	Bartell (2021)	Two studies	Meta-analysis	Varied by study	Varied by study	Varied by study	PFOA exposure	Testicular cancer incidence	Varied by study	Increase in cancer risk per 10 ng/mL increase in serum PFOA = 3% (2%, 4%)
Hypothyroidism										
PFOA~	Kim (2018)	Seven studies after excluding outliers	Cross-sectional	2007–2009	USA	Adults > 20 years old	PFOA in serum	Subclinical hypothyroidism in women	Age, race, drinking, smoking, and natural log-urinary iodine	7.42 (1.14–48.12) OR of subclinical hypothyroidism risk per ln-unit ng/mL increase in serum PFOA
PFOS*	Wen (2013)	1181 individuals	Cross-sectional	2007–2009	USA	Adults > 20 years old	PFOS in serum	Subclinical hypothyroidism in women	Age, race, drinking, smoking, and natural log-urinary iodine	3.03 (1.14–8.07) OR of subclinical hypothyroidism risk per ln-unit ng/mL increase in serum PFOA
Adult obesity										
PFOS*	Liu (2018a)	520 individuals	Randomized clinical trial	2003–2007	Boston, MA and Baton Rouge, LA	Over-weight and obese 30–70-year olds	PFOS in serum	Body weight	Age, sex, race, baseline BMI, education, smoking status, alcohol consumption, physical activity, and dietary intervention group	Higher baseline levels of PFOS associated with greater weight regain (1.5 ± 0.6–3.2 ± 0.6 kg)

Table 1 (continued)

Exposure	Author (Year)	N	Study type	Date of recruitment	Location	Population	Exposure	Outcome	Covariates	Results
Adult-onset type 2 diabetes										
PFOA*	Sun (2018)	1586 females	Prospective nested case–control study	1995–2000	USA	Female nurses 32–52 years old	PFAS in plasma	T2D	Age, month, and fasting status at sample collection and menopausal status and hormone replacement therapy	1.54 (1.04–2.28) OR of T2D in highest tertile of exposure compared to lowest
Gestational diabetes										
PFOA*	Zhang (2015)	258 females	Prospective cohort	2005–2009	Michigan and Texas	Women discontinuing contraception to become pregnant	PFOA in serum	GDM	Age, BMI, and parity conditional on gravidity	1.61 (1.05–2.49) OR of GDM per SD increment of PFOA exposure
Endometriosis										
PFOA*	Buck Louis (2012)	626 females	Prospective nested case–control study	2007–2009	Salt Lake City, UT and San Francisco, CA	Women 18–44 years old	PFOA in serum	Endometriosis	Age, BMI, and parity	1.89 (1.17–3.06) OR of endometriosis per log unit exposure of PFOA
Polycystic ovarian syndrome										
PFOA*	Vagi (2014)	102 females	Case–control	2007–2008	Los Angeles, CA	Women 18–45 years old	PFOA in serum	PCOS	Age, BMI, and race	6.93 (1.79–29.92) OR of PCOS in highest tertile compared to lowest
PFOS*	Vagi (2014)	102 females	Case–control	2007–2008	Los Angeles, CA	Women 18–45 years old	PFOS in serum	PCOS	Age, BMI, and race	5.79 (1.58–24.12) OR of PCOS in highest tertile compared to lowest
Couple infertility										
PFOA*	Bach (2015)	1601 females	Case–control	1996–2002	Denmark	Pregnant women	PFOA in serum	Time to pregnancy	Age, socio-economic status, BMI, and parity	0.67 (0.51–0.88) fecundability ratio per log unit PFOA exposure

Table 1 (continued)

Exposure	Author (Year)	N	Study type	Date of recruitment	Location	Population	Exposure	Outcome	Covariates	Results
PFOS*	Bach (2015)	1601 females	Case–control	1996–2002	Denmark	Pregnant women	PFOS in serum	Time to pregnancy	Age, socio-economic status, BMI, and parity	0.62 (0.47–0.83) fecundability ratio per log unit PFOS exposure
Breast cancer										
PFOA*	Wielsoe (2017)	161 females	Case–control	2000–2003, 2011–2014	Greenland	Inuit women	PFOA in serum	Breast cancer	Age, BMI, cotinine levels, parity, and breastfeeding	1.26 (1.01–1.58) OR of breast cancer with PFOA exposure
PFOS*	Wielsoe (2017)	161 females	Case–control	2000–2003, 2011–2014	Greenland	Inuit women	PFOS in serum	Breast cancer	Age, BMI, cotinine levels, parity, and breastfeeding	1.02 (1.01–1.03) OR of breast cancer with PFOS exposure
Pneumonia										
PFOA*	Impinen (2019)	1270 females	Cohort	1999–2008	Norway	Children	PFOA in maternal serum	Number of infections by age 3	Maternal age, maternal BMI, maternal education, parity, and smoking during pregnancy	1.27 (1.12–1.43) RR of bronchitis/pneumonia
PFOS*	Impinen (2019)	1270 females	Cohort	1999–2008	Norway	Children	PFOS in maternal serum	Number of infections by age 3	Maternal age, maternal BMI, maternal education, parity, and smoking during pregnancy	1.20 (1.07–1.34) RR of bronchitis/pneumonia

*Sensitivity analysis

~From meta-analyses

Table 2 Exposure–response relationships

Outcome	Exposed population	Exposure	Exposure modeling (ng/mL)	ERR	Source of ERR
Low birth weight~					
	Females 18–49	PFOA	Continuous	$\beta = -10.5$ g/PFOA (ng/mL)	Steenland (2018)
		PFOS	Continuous	$\beta = -45.2$ g/doubling of PFOS (ng/mL)	Meng (2018)
Childhood obesity at age 10~					
	Females 18–49	PFOA	Continuous	$\beta = 0.09$ per PFOA (ng/mL)	Liu (2018a)
		PFOS	Continuous	$\beta = 0.18$ per ln((PFOS) (ng/mL))	Lauritzen (2018)
Kidney cancer					
	Adults 18+	PFOA	Continuous	OR = 1.16 per 10 ng/mL PFOA	Bartell (2021)
Testicular cancer					
	Males 18+	PFOA	Continuous	OR = 1.03 per 10 ng/mL PFOA	Bartel (2021)
Hypothyroidism					
	Females 18–49	PFOA	Continuous	OR = 7.42 per ln(PFOA (ng/mL))	Wen (2013)
		PFOS	Continuous	OR = 3.03 per ln(PFOA (ng/mL))	
Adult obesity					
	Adults 18+	PFOS	Tertile 1: < 19.2	$\beta = 1.5$ kg gained	Liu (2018a)
			Tertile 2: 19.2–32.1	$\beta = 3.5$ kg gained	
			Tertile 3: > 32.1	$\beta = 3.2$ kg gained	
Adult-onset type 2 diabetes					
	Females 18–49	PFOA	Tertile 1: < 3.76	OR = 1	Sun (2018)
			Tertile 2: 3.76–5.48	OR = 1.27	
			Tertile 3: > 5.48	OR = 1.54	
Gestational diabetes					
	Females 18–49	PFOA	Continuous	When ln(1 + PFOA (ng/mL) increases by 1 SD, OR increases by 1.61	Zhang (2015)
Endometriosis					
	Females 18–49	PFOA	Continuous	OR = 1.89 per log((PFOA (ng/mL))	Buck Louis (2012)
Polycystic ovarian syndrome					
	Females 15–45	PFOA	Tertile 1: < 2.6	OR = 1	Vagi (2014)
			Tertile 2: 2.6–4.1	OR = 1.65	
			Tertile 3: > 4.1	OR = 6.93	
		PFOS	Tertile 1: < 6.2	OR = 1	
			Tertile 2: 6.2–8.6	OR = 3.43	
			Tertile 3: > 8.6	OR = 5.79	
Couple infertility					
	Females 18–49	PFOA	Continuous	OR = 0.67 per log((PFOA (ng/mL))	Bach (2015)
		PFOS	Continuous	OR = 0.62 per log((PFOS (ng/mL))	
Breast cancer					
	Females 18–49	PFOA	Continuous	OR = 1.26 per PFOA (ng/mL)	Wielsoe (2017)
		PFOS	Continuous	OR = 1.02 per PFOS (ng/mL)	
Pneumonia~					
	Females 18–49	PFOA	Continuous	RR = 1.27 per PFOA (ng/mL)	Impinen (2019)
		PFOS	Continuous	RR = 1.20 per PFOS (ng/mL)	

~ Serum concentrations in females of childbearing age was used as a proxy for prenatal exposure

The AF of a risk factor can be defined as the proportional decrease in the number of cases of ill health or deaths as a result of reducing the risk factor to a reference level and can be estimated using the following equation:

$$\mathrm{AF} = \mathrm{Prevalence}_{\mathrm{exposure}} * (\mathrm{relative\ risk(RR)} - 1) \Big/ \left[1 + \left(\mathrm{prevalence}_{\mathrm{exposure}} * (\mathrm{RR} - 1)\right)\right], \tag{3}$$

where RR represents the risk of morbidity associated with the specific exposure relative to the reference level (Levin 1953).

The first step in calculating attributable disease burden was to quantify exposure. We focused our analysis on PFOA and PFOS, as they are the most widely studied members of the PFAS class and evidence of their health effects is strongest. Because these two chemicals co-occur, as a conservative measure, we calculated disease burden based on PFOA and PFOS separately as proxies for long-chain PFAS exposure. Our source for distributions of exposure was the 2017–2018 cycle of NHANES, as this contains the most recent nationally representative data. For each analysis, we focused on the relevant subsection of the population (e.g., women of childbearing age when considering PCOS). For childhood obesity we used data from the 2007–2008 cycle to quantify in utero exposure among children who were age 10 in 2017–2018. NHANES measured serum concentrations of PFOA and PFOS with online solid-phase extraction coupled with high-performance liquid chromatography–tandem mass spectrometry; an extensive methodology is provided in the NHANES Laboratory Procedures Manual (CDC 2016) Main estimates used PFOA levels, while sensitivity analyses considered PFOS levels, as well.

We stratified the US population into percentile groupings of serum PFAS concentration (< 10th, 10th–24th, 25th–49th, 50th–74th, 75th–89th, 90th–99th, and > 99th). As a conservative measure, we assumed exposures within each percentile grouping to be at the lowest end of the range (e.g., corresponding to the 10th, 25th, 50th, 75th, 90th, and 99th percentile) and assumed no exposure for the lowest 10% of the population, our reference group (Table 3).

Once we established the exposure level across each percentile group, we calculated increments in disease or disability over the baseline population rate due to exposure. 2018 US Census estimates (USC Bureau 2020) were used to convert the baseline prevalence or incidence values to the appropriate population size (subsequent sections identify the sources of prevalence/incidence data for each outcome). We then applied the previously selected ERRs to quantify attributable burdens of disease within each group. If an ERR was based on continuous exposure in the literature, main analyses employed a reference level of 0.1 ng/mL below which no effects were assumed to be observed. If an ERR was based on tertiles or quartiles of exposure in the literature, we used the lowest quantile as the reference level (Table 3). ORs were converted to RRs to avoid overestimation following published practice (Knol et al. 2012), and Levin's formula was used to tabulate AFs based on RRs (Levin 1953).

Once we estimated the increase in cases attributable to PFOA/PFOS exposure for the 13 outcome measures, we calculated associated economic costs using available data on cost per case, derived from previously published estimates of direct and/or indirect healthcare and societal costs, and the size of the population at risk (Supplementary Tables 1–13). All cost estimates were adjusted to reflect the annual average for 2018 in US dollars using the All Items Consumer Price Index (US Bureau of Labor Statistics 2020).

Table 3 NHANES exposure data

Outcome	Exposed population	NHANES years of exposure	Exposure chemical	Concentrations assigned to each percentile of exposure						
				0–9	10–24	25–49	50–74	75–89	90–99	> 99
Low birth weight*, adult diabetes, gestational diabetes, endometriosis, couple infertility, breast cancer, pneumonia*, and hypothyroidism										
	Females 18–49	2017–2018	PFOA (ng/mL)	0.00	0.47	0.67	0.97	1.47	2.37	5.17
			PFOS (ng/mL)	0.00	1.10	1.70	2.60	3.90	5.70	11.9
Childhood obesity at age 10*										
	Females 18–49	2007–2008	PFOA (ng/mL)	0.00	2.30	2.70	3.50	4.60	5.90	7.10
			PFOS (ng/mL)	0.00	4.70	7.20	9.90	17.70	24.30	32.20
Adult obesity and kidney cancer										
	Adults 18+	2017–2018	PFOA (ng/mL)	0.00	0.67	0.97	1.47	2.17	3.07	8.30
			PFOS (ng/mL)	0.00	1.60	2.70	4.70	7.80	12.0	26.2
Polycystic ovarian syndrome										
	Females 15–45	2017–2018	PFOA (ng/mL)	0.00	0.47	0.67	0.87	1.37	2.07	5.17
			PFOS (ng/mL)	0.00	1.10	1.70	2.50	3.60	5.30	10.7
Testicular cancer										
	Males 18+	2017–2018	PFOA (ng/mL)	0.00	0.87	1.17	1.67	2.27	3.27	8.30
			PFOS (ng/mL)	0.00	2.60	3.60	5.80	9.20	13.0	26.2

*Serum concentrations in females of childbearing age were used as a proxy for prenatal exposure

Table 4 Total disease burden and costs in 2017–2018

	Primary analysis	Sensitivity analysis	
	Main estimate from meta-analyses	Low estimate	High estimate
Low birth weight			
Attributable incident cases per year	10,053	–	96,847
Attributable fraction	3.17%	–	30.7%
Total cost per annual incident case	$1,420,000,000	–	$13,700,000,000
Direct cost of hospitalization	$305,000,000	–	$2,940,000,000
Indirect cost due to lost IQ points	$1,110,000,000	–	$10,700,000,000
Childhood obesity at age 10			
Attributable incident cases per year	127,362	–	462,119
Attributable fraction	3.78%	–	13.70%
Incremental lifetime medical cost of an obese child relative to normal weight child due to annual incident cases	$2,650,000,000	–	$9,600,000,000
Kidney cancer			
Attributable incident cases per year	142	–	–
Attributable fraction	0.34%	–	–
Total cost per annual incident case	$184,000,000	–	–
Direct medical cost during 1st year of diagnosis	$4,740,000	–	–
Indirect cost as DALY lost over 10 years	$180,000,000	–	–
Testicular cancer			
Attributable incident cases per year	5	–	–
Attributable fraction	0.076%	–	–
Total cost per annual incident case	$6,850,000	–	–
Direct medical cost of treatment	$139,000	–	–
Indirect cost as DALY lost over 10 years	$6,710,000	–	–
Hypothyroidism in females			
Attributable incident cases per year	14,572	–	59,939
Attributable fraction	5.0%	–	20.7%
Total cost per annual incident case	$1,260,000,000	–	$5,180,000,000
Direct cost of new cases of hypothyroidism annually	$42,100,000	–	$173,000,000
Indirect cost as DALY lost over 10 years	$1,220,000,000	–	$5,000,000,000
Adult obesity			
Attributable incident cases per year	–	4,294,379	–
Attributable fraction	–	2.98%	–
Total 15-year cost per annual incident case	–	$17,000,000,000	–
Direct medical cost for newly obese 35-year olds	–	$3,210,000,000	–
Indirect cost of QALY lost over 15 years	–	$13,800,000,000	–
Adult type II diabetes in females			
Attributable incident cases per year	–	1728	–
Attributable fraction	–	0.37%	–
Lifetime cost of treating type II diabetes and associated complications due to annual incident cases	–	$140,000,000	–
Gestational diabetes			
Attributable incident cases per year	–	6061	12,474
Attributable fraction	–	2.85%	5.87%
Total cost per annual incident case	–	$414,000,000	$852,000,000
Direct medical cost	–	$73,300,000	$150,000,000
Indirect cost of lost productivity from adverse birth effects	–	$341,000,000	$702,000,000
Endometriosis			

Table 4 (continued)

	Primary analysis	Sensitivity analysis	
	Main estimate from meta-analyses	Low estimate	High estimate
Attributable incident cases per year	–	696	18,062
Attributable fraction	–	0.43%	11.27%
Total 10-year cost per annual incident case	–	$397,000,000	$10,200,000,000
Direct medical cost over 10 years	–	$21,100,000	$547,000,000
Indirect cost as DALY lost over 10 years	–	$376,000,000	$9,760,000,000
Polycystic ovarian syndrome			
Attributable incident cases per year	–	7209	7505
Attributable fraction	–	5.92%	6.16%
Annual cost of initial PCOS evaluation and treatment of comorbidities due to annual incident cases	–	$10,500,000	$10,900,000
Couple infertility			
Attributable cases of ART SET utilized per year	–	593	26,160
Attributable fraction	–	0.25%	10.86%
Cost of attributable ART SET utilization per annual incident case	–	$37,600,000	$1,660,000,000
Breast cancer			
Attributable incident cases per year	–	421	3095
Attributable fraction	–	0.50%	3.65%
Total 10-year cost per annual incident case	–	$555,000,000	$4,080,000,000
Direct medical cost for 6 months following diagnosis per annual incident cases	–	$21,700,000	$159,000,000
Indirect cost as DALY lost over 10 years	–	$533,000,000	$3,920,000,000
Pneumonia			
Attributable incident cases per year in children 0–3 years old	–	447	6759
Attributable fraction	–	0.58%	8.81%
Total cost per incident case of pneumonia in 0–3-year olds	–	$1,490,000	$22,500,000
Direct medical cost of case across all healthcare settings	–	$1,320,000	$20,000,000
Indirect cost of parental absenteeism	–	$166,000	$2,510,000
Total cost	$5.52 billion	($5.52 billion–$62.6 billion)	

Sensitivity Analyses

As our main result, we reported the PFOA disease burden and cost estimates for the five disease outcomes with meta-analytic associations and then summed them. We then generated alternative estimates through multiway sensitivity analyses to provide the most accurate range of possible costs (Table 4). First, we calculated disease burden and cost estimates using ERRs for PFOS and serum levels from NHANES for the same group of outcomes. We then calculated disease burden and cost estimates for PFOA using an expanded group of outcomes that included both those conditions for which there were meta-analytic results and those for which there were results from systematic or scoping reviews. We also examined the influence of a higher reference level (1.0 ng/mL) on disease burden and costs for which ERRs were based on continuous exposure. Finally, we repeated this analysis for the expanded group of conditions, substituting PFOS for PFOA. The boundaries of the sensitivity analysis were identified using the lowest and highest values for each of the adverse endpoints studied, which were aggregated to create a range for probable disease costs due to PFAS.

The following sections elaborate details of our methods specific to each disease outcome.

Low Birth Weight

We updated a previously published approach to quantifying PFAS-attributable LBW (Malits et al. 2018) to include new literature and an estimate for PFOS, which we used in a sensitivity analysis. Briefly, we compared observed LBW in 2017–2018 to LBW in a counterfactual scenario in which PFOA/PFOS-attributable reductions in birth weight were eliminated, with the difference representing PFOA/PFOS-attributable LBW. For each 1.0 ng/mL of PFOA exposure above 0.1 ng/mL, a 10.5 g decrease [95% confidence interval

(CI) − 16.7, − 4.4] in birth weight was applied in main analyses, based on the results of an updated meta-analysis (Steenland et al. 2018). Sensitivity analyses applied the lower 3.3 g decrement identified in a subset of studies with later pregnancy measures. For PFOS, we applied a 45.2 g decrease (95% CI − 76.8, − 13.6) in birth weight per doubling of early pregnancy maternal plasma concentrations from a study of 3535 mother–infant pairs in the Danish National Birth Cohort study (Meng et al. 2018). We used natality data from the National Vital Statistics System of the National Center for Health Statistics (CDC/NCHS 2014, 2018) to determine the actual mean birth weight, total number of births, and number of LBW births for 2017 and 2018 (the exposed scenario) and then increased mean birth weight in each PFOA/PFOS centile by the absolute value of the attributable decrement to calculate the number of LBW births in a scenario free of PFOA/PFOS effects. The PFOA/PFOS-attributable LBW disease burden was the difference in LBW births between the two, assuming a normal distribution of birth weight (Table S1). The average of results for 2017 and 2018 was calculated to represent PFAS-attributable LBW in 2018.

We calculated the total cost of LBW attributable to in utero PFAS exposure by adding the LBW-associated costs of hospitalization for medical concerns (direct cost) to the lost lifetime economic productivity, operationalized as loss of IQ points due to LBW (indirect cost). The direct cost of hospitalization was estimated at $30,364 per case in 2018 (Kowlessar et al. 2011). LBW has been associated with a 4.98 point loss in IQ (95% CI 3.20, 6.77) (Kormos et al. 2014). Applying a 3% discount rate for lifetime earnings, each IQ point loss was valued at $22,190 in 2018 (Gould 2009; Max et al. 2004). Both costs were multiplied by the number of additional LBW babies born over the 2 years attributable to PFOA and PFOS exposure to get the total cost.

Childhood Obesity

We first quantified changes in body mass index (BMI) Z-score in subpopulations of children with increasing prenatal PFOA exposure by applying results of a meta-analysis of ten cohort studies, which identified a 0.09 increase in BMI Z-score for each ng/mL increase in PFOA (Liu et al. 2018b). For PFOS, we utilized a cohort study of 412 Norwegian and Swedish mother–infant pairs in which a 1 ng/mL increase in maternal serum levels was associated with 0.18 increase in BMI Z-score (Lauritzen et al. 2018). A 0.1 ng/mL reference level was applied in all analyses, below which no effects on BMI Z-score were included. The distribution of PFOA and PFOS in US women age 18–49 years in 2007–2008 was used as a proxy for the distribution in pregnant women during that time period. To estimate increases in childhood obesity in 10-year olds due to prenatal PFOA/PFOS exposure, we calculated increases in BMI Z-score and quantified incremental increases in $Z > 1.64$ (95th percentile).

Incremental increases in obesity were calculated from PFOA/PFOS-attributable increases in BMI Z-score using the NORMDIST function in Excel, assuming a mean = 0 and standard deviation (SD) = 1 without exposure. Increases in percent obese individuals were then multiplied by the number of 10-year olds in 2018 identified in US Census population estimates (Table S2) (USC Bureau 2020; Hales et al. 2017). We calculated the economic burden of PFOA/PFOS-attributable cases of childhood obesity based on an estimated lifetime medical cost of childhood obesity at age 10 of $20,780 in 2018 dollars (Finkelstein et al. 2014).

Kidney Cancer

We utilized the ERR from Bartell et al. to identify the PFOA-attributable increased odds of kidney cancer based on a pooled increased risk of 16% per 10 ng/mL of PFOA exposure from a meta-analysis of four papers demonstrating the link between PFOA and kidney cancer (Bartell and Vieira 2021). We calculated ORs for PFOA in each percentile grouping based upon exposure levels in NHANES 2017–2018, applying a reference level of 0.1 ng/mL below which we assumed there was no increase in odds of kidney cancer. We then converted the ORs to RRs and adjusted for a kidney cancer prevalence of 12.89 per 10,000 adults in the US (Surveillance Research Program Surveillance, Epidemiology, and End Results Program). Afterward, we weighted the RRs by exposure percentile to calculate the AFs across all exposure percentiles using Levin's equation (Levin 1953). The population incidence of kidney cancer, 16.9 per 100,000 adults/year, was obtained from the Surveillance, Epidemiology, and End Results Program and multiplied by the AF across the modeled range of population exposures and the US Census population estimates of the annual average number of adults over age 18 years in 2018 to quantify incident cases of kidney cancer attributable to PFOA exposure (Table S3) (Surveillance Research Program Surveillance, Epidemiology, and End Results Program).

Each case of newly diagnosed kidney cancer was associated with direct medical expenses of $33,485 in the first year alone in 2018 (Shih et al. 2019). We multiplied the cost by the PFOA-attributable cases for a total direct cost of first-year medical expenses for newly diagnosed kidney cancer in American adults. We then calculated the indirect 10-year cost of kidney cancer as lost disability-adjusted life years (DALY, 0.288 for each year, valued at $50,000/year) over 10 years, discounting 3% per year for future preference (Neumann et al. 2014). The total 10-year cost for a case of kidney cancer is the sum of first-year medical expenses and accrued indirect costs (DALY loss).

Testicular Cancer

Similarly to kidney cancer, we utilized an ERR from Bartell et al. that demonstrated a 3% increase in risk of testicular cancer per 10 ng/mL of PFOA exposure from a meta-analysis of two studies (Bartell and Vieira 2021). The same methodology was applied as with kidney cancer to determine ORs, convert to RRs using a prevalence of 0.0817% of adult males, and identify a weighted AF (US Cancer Statistics Working Group 2020). The AF was then multiplied by the US population of adult males and baseline incidence of testicular cancer of 5.7 per 100,000 to identify the PFOA-attributable cases of testicular cancer in 2018 (Table S4) (Surveillance Research Program Surveillance, Epidemiology, and End Results Program). The number of PFOA-attributable testicular cancer cases was multiplied by \$26,236, the estimated cost of each new case to the US healthcare system in 2018 dollars (Aberger et al. 2014). As with kidney cancer, we calculated the indirect 10-year cost of testicular cancer as lost DALY (0.288 for each year, valued at \$50,000/year) over 10 years, discounting 3% per year for future preference, which we then summed with the direct cost of a new case of testicular cancer (Neumann et al. 2014).

Hypothyroidism

Wen et al.'s analysis based on 2007–2010 NHANES data from 1181 adults provided us with an OR of 7.42 (95% CI 1.14–48.12) to estimate the increase in subclinical hypothyroidism per ln-unit increase of PFOA serum concentration (Wen et al. 2013). We conducted a sensitivity analysis using the association between PFOS and increased odds of subclinical hypothyroidism in females from the same study (OR 3.03; 95% CI 1.14–8.07).

For both PFAS, we applied the OR to our exposure percentiles of PFOA/PFOS in adult women then converted to an RR using a prevalence of clinical hypothyroidism of 0.3% (Hollowell et al. 2002). A weighted AF was then calculated and multiplied by the US population of adult women and incidence of subclinical hypothyroidism of 226.2 per 100,000 adults to obtain PFOA-attributable cases of subclinical hypothyroidism (Garmendia Madariaga et al. 2014). This was adjusted downward by 0.3% to account for the baseline prevalence of hypothyroidism (Table S5) (Hollowell et al. 2002).

The annual direct medical cost per case of hypothyroidism is valued at \$2555 with associated \$171 in indirect costs due to lost productivity in 2015 (Hepp et al. 2021). We converted the sum of these costs into 2018 dollars (\$2888) and multiplied by the PFAS-attributable cases for a total annual cost of subclinical hypothyroidism in adult females. Given the variable clinical course of hypothyroidism, we chose to calculate costs for a single year of treatment due to PFAS rather than lifelong costs. As hypothyroidism is a chronic disease, we modeled an indirect 10-year cost as lost DALY (0.019 for each year, valued at \$50,000/year) over 10 years, discounting 3% per year for future preference (Neumann et al. 2014).

Adult Obesity

To quantify PFOS-attributable adult obesity, we modeled increases in obesity by shifting the mean BMI for US adults age > 18 years in relation to PFOS exposure in each centile and estimated increases in percentages of the population with BMI > 30 kg/m^2. We applied results from Liu et al.'s study of 520 adults followed for 6–24 months after the cessation of a 2-year clinical trial of energy-restricted diets on weight change that reported those with PFOS levels > 32.1 ng/mL gained 3.2 kg over the 6–24-month study period, those with levels 19.2–32.1 ng/mL gained 3.5 kg, and those with levels < 19.2 ng/mL gained 1.5 kg (Liu et al. 2018a). Weight gain across tertiles was then linearized across the percentiles to estimate a finer distinction between those with varying exposures (Table S6, Table S14, Fig. S1).

After applying NHANES 2017–2018 PFOS levels to calculate attributable annual weight gain, the additional weight was added to mean weight in the unexposed scenario, as calculated from mean BMI (29.78 kg/m^2) and height (1.66 m) and an exposed mean BMI was calculated from the new weight and same height. Increases in obesity (BMI > 30 kg/m^2) in each exposed subpopulation were calculated by subtracting the percent obese in the exposed scenario to the unexposed counterfactual. The increase in obesity was multiplied by the annual number of adults in the US in 2018 as estimated by the US Census and adjusted for a baseline prevalence of obesity (42.4%) to obtain the number of cases of incident obesity among adults over the age of 18 attributable to PFOS (Table S6) (Fryar et al. 2016; Hales et al. 2020).

We estimated the long-term cost of obesity as a sum of the 15-year direct annual medical cost of obesity (e.g., medical expenses) and the indirect cost of quality-adjusted life years (QALY) lost, using a single age group as a model. We selected 35-year olds, as obesity rates increase with age and this age cohort would allow us to model a 15-year period with the assumption that the majority of 35-year-old obese individuals will remain obese and continue to live for at least 15 years. Using the annual direct medical cost of adult obesity as \$2741 in 2005 dollars and discounting for future preference (3% annually), we calculated that a 35-year old who became obese as a result of PFAS exposure in 2017–2018 would incur \$43,334 in direct medical costs over 15 years (Cawley and Meyerhoefer 2012). This cost was multiplied by the incremental increase in obesity and the total population of 35-year olds in the US. The indirect cost of adult obesity due to PFAS was calculated as QALY lost due to obesity, with each QALY assigned a value of \$50,000 (Eq. 4) (Muennig et al. 2006; Neumann et al. 2014). Results for males and females were calculated separately, as QALY lost to obesity are sex specific (4.4 years for men and 7.2 years for women), and the final indirect costs for each PFAS of both genders were summed (Muennig et al. 2006).

$$\text{Indirect cost} = \frac{\$50,000 \times \text{populationobese} \times \text{QALY}}{\left(1.03^{15}\right)}. \quad (4)$$

Type 2 Diabetes

We extrapolated incident cases of T2D in 2017–2018 due to PFOA exposure in females over age 18 years using the findings of a case–control study of 1586 women nested within the Nurses' Health Study II that found higher odds of T2D associated with each tertile increase in PFOA concentration (Sun et al. 2018). Odds of incident T2D were linearized across tertiles to estimate a finer distinction with varying exposures as described for adult obesity. We converted the ORs to RRs and then applied Levin's equation to calculate AFs from the RRs (Levin 1953). For each exposed subpopulation, the calculated AF was multiplied by the incidence rate of T2D [6.9 per 1000 American adults (CDC 2020)] and the annual population of US women in 2018. To avoid overestimation, we adjusted for baseline prevalence of T2D (13.0%) to obtain a final estimate of PFOA-attributable cases of T2D in adult women (Table S7) (CDC 2020). The lifetime cost of T2D was estimated at \$93,183 per individual in 2018 and multiplied by the number of PFOA-attributable cases in 2018 (Zhou et al. 2013).

Gestational Diabetes

We applied findings from a prospective cohort study of 501 women in whom preconception serum PFOA levels were associated with GDM (OR 1.61 per 0.43 SD increase in PFOA concentration; 95% CI 1.14–3.02) (Zhang et al. 2015). We assumed levels of PFOA exposure among women age 18–49 years in 2017–2018 NHANES to be similar to those in pregnant women of the same year and applied a reference level of 0.1 ng/mL below which we assumed no effect. As with prior calculations, we converted ORs to RRs and then applied Levin's equation to calculate AFs from the RRs (Levin 1953). The AF across all centiles was multiplied by the number of births in 2017–2018 and the prevalence rate of GDM (5.60%) to estimate the annual PFOA-attributable cases of incident GDM (assuming that the prevalence of GDM is the same as the incidence, as the natural progression of the disease is < 1 year) (Table S8) (CDC/NCHS 2014).

Each case of GDM was estimated to have an annual medical cost of \$12,089 and lifetime cost due to lost productivity for adverse birth outcomes associated with GDM of \$56,237 in 2018 dollars (Peterson et al. 2015). These costs were multiplied by the number of PFAS-attributable cases of GDM.

Endometriosis

After determining the percentile groupings of serum PFOA levels in women age 18–49, we utilized ORs for associations between PFOA and endometriosis from the Endometriosis: Natural History, Diagnosis, and Outcomes (ENDO) study, a case–control study of 495 women age 18–44 years that found an association between serum PFOA levels and higher odds of endometriosis (Buck Louis et al. 2012). We calculated ORs for PFOA in each percentile grouping based upon exposure levels in NHANES 2017–2018, applying a reference level of 0.1 ng/mL below which we assumed there was no increase in odds of endometriosis. We converted the ORs to RRs and applied Levin's equation to calculate AFs as with prior outcomes (Levin 1953). The population incidence of endometriosis, 237 per 100,000 women/year, was obtained from the Nurses' Health Study and multiplied by the AF across the modeled range of population exposures and the US Census population estimates of the number of women age 18–49 years in 2018 and then adjusted for baseline prevalence (6.1%) (Fuldeore and Soliman 2017) to quantify incident cases of endometriosis attributable to PFOA exposure (Table S9) (Missmer et al. 2004).

Following the methodology of Attina et al. (2016), we modeled the direct cost of endometriosis as the total healthcare costs over 10 years of treatment, valued at \$30,292 in 2018 dollars (Fuldeore et al. 2015). We also calculated the indirect cost of endometriosis by aggregating lost DALY (0.123 for each year with endometriosis, valued at \$50,000/year) over 10 years, discounting 3% per year for future preference (Neumann et al. 2014). These costs were multiplied by the newly incident cases of endometriosis attributable to annual PFOA exposure to obtain the annual PFOA-attributable economic burden.

Polycystic Ovarian Syndrome

We quantified incident cases of PCOS in women age 15–45 years attributable to PFOA/PFOS by applying ERRs from a case–control study by Vagi et al. of 52 PCOS patients and 50 controls in Los Angeles to our percentile groupings of PFOA and PFOS exposure (Vagi et al. 2014). Linearized ORs were calculated as described for adult obesity, and ORs for exposure percentile groups were assigned based on the corresponding tertiles of exposure identified in the Vagi et al. study. The ORs for the second and third tertiles versus the first were 1.65 and 6.93 for PFOA ($p_{\text{trend}} = 0.003$) and 3.43 and 5.79 for PFOS ($p_{\text{trend}} = 0.005$), respectively (Vagi et al. 2014). ORs were converted to RRs, which were then converted to AFs using Levin's equation, (Levin 1953) multiplied by the incidence of PCOS (2 per 1000 women based on a study of PCOS incidence in the United Kingdom) (Ding 2017) and the population of women age 15–45 years in the US (USCBureau 2020), and adjusted for baseline cases of PCOS (6.6%) (Azziz et al. 2004) to calculate the number of PFOA/PFOS-attributable cases of PCOS in 2018 (Table S10).

The annual medical cost of PCOS in the US was estimated as $4.37 billion for 4 million women or $1092 per PCOS case in 2004 dollars (Azziz et al. 2005). This cost estimate includes the annual cost of initial evaluation and treatment of associated menstrual dysfunction, infertility, T2D, and hirsutism. The cost per case was then multiplied by the PFOA/PFOS-attributable cases of PCOS and adjusted to 2018 dollars ($1452 per case) to determine the annual economic burden due to PCOS-related healthcare visits.

Couple Infertility

We quantified PFAS-attributable cases of couple infertility, defined as TTP > 12 months, based on exposure data from 2017 to 2018 NHANES in women of childbearing age (age 18–49 years). To calculate the OR for infertility in each exposure group, we leveraged data from a case–control analysis of 910 women nested within the Norwegian Mother and Child Cohort Study (Whitworth et al. 2012). Although TTP is a couple-based outcome, chemical exposures were measured only in women, a common limitation among TTP studies. We calculated a linearized OR for estimated serum PFOA/PFOS in each exposure group and assigned an OR for infertility based on the corresponding quartile from the Norwegian study. As with prior estimates, we converted the OR to a RR based on a prevalence rate of impaired fecundity (13.1%) (CDC 2018) and subsequently calculated AFs using Levin's equation (CDC 2018; Levin 1953). We multiplied the AFs by the incidence of infertility in 2018 (63.6 per 10,000 women) and the US population of women age 18–49 years and then adjusted for the baseline prevalence of infertility (13.1%) to quantify attributable cases of infertility (Table S11) (Boivin et al. 2007; Stahlman and Fan 2019).

We applied a 56% utilization rate of assisted reproductive technologies (ART) among infertile couples to assess cost (Boivin et al. 2007). The cost of a single fresh cycle of ART was valued at $63,530 in 2018 dollars. This cost is inclusive of direct maternal and infant costs from 27 weeks prior to delivery through the first year of an infant's life and accounts for the increased rate of multiparity and premature births associated with ART (Crawford et al. 2016). This cost was multiplied by the PFOA/PFOS-attributable annual use of ART in 2018 to estimate the total cost.

Breast Cancer

We applied an OR of 1.26 per ng/mL of PFOA and 1.02 per ng/mL of PFOS from a case–control study of 161 Inuit women in Greenland (Wielsoe et al. 2017). We then calculated the ORs for PFOA/PFOS-associated breast cancer for each of our exposure centiles by multiplying the ORs from this study by the levels of exposure from 2017 to 2018 NHANES among women age 18–49, assuming a reference level of 0.1 ng/mL below which we modeled no effect. The OR for each centile was converted to an RR, which was further transformed into an AF. PFOA/PFOS-attributable cases of breast cancer were then determined by multiplying the weighted AFs by the population of women age 18–49 years and the US breast cancer incidence rate (125.1 per 100,000) and then adjusting for a baseline prevalence of 1.2% (Table S12) (US Cancer Statistics Working Group 2020).

The PFOA/PFOS-attributable cases of female breast cancer were multiplied by the healthcare costs for the first 6 months of a new breast cancer diagnosis. While there are varied lifetime cost estimates of having breast cancer depending on the different stages at which patients are diagnosed, $51,498 in 2018 dollars is the minimum estimated cost a patient will incur throughout the first 6 months of diagnosis regardless of prognosis or odds of remission (Lamerato et al. 2006). As with kidney and testicular cancer, we calculated the indirect 10-year cost of breast cancer as lost DALY (0.288 for each year, valued at $50,000/year) over 10 years, discounting 3% per year for future preference (Neumann et al. 2014).

Pneumonia

To determine the PFAS-attributable increase in pneumonia infections among children age < 3 years, we utilized RRs of 1.27 (95% CI 1.12–1.43) and 1.20 (95% CI 1.07–1.34) for PFOA and PFOS, respectively, from an analysis of 1270 maternal-child pairs in the Norwegian Mother and Child Cohort Study (Impinen et al. 2019). We applied the RR to each percentile grouping of maternal serum PFOA/PFOS levels to calculate the increased risk of pneumonia and bronchitis among children age < 3 years as a result of in utero exposure to PFAS. The RR was then transformed to a weighted AF across all centiles using Levin's equation (Levin 1953) and multiplied by the US population of children age < 3 years and a weighted average incidence rate of 49.4 per 10,000 children, as derived from the incidence rates of pneumonia in children < 2 years old and 2–4 years old, to obtain the PFOA/PFOS-attributable cases of pneumonia in children under age 3 (Table S13) (Jain et al. 2015).

The economic burden of pneumonia in children age < 3 years was constructed as a combination of the overall direct cost of a pneumonia episode (emergency room visit, hospitalization, or outpatient treatment) and the indirect cost defined as lost parental weekly earnings. The average cost across all healthcare settings per case of pneumonia was determined to be $2952 in 2018 dollars (Tong et al. 2018). For the indirect cost, we multiplied the mean weekly earnings of full-time wage and salary workers in 2014 ($113 per diem) by the average length of stay for a pneumonia hospitalization (3.1 days) to obtain an indirect cost of $350 in lost parental earnings per case or $372 in 2018 dollars (US Bureau of Labor Statistics 2021; Williams et al. 2018). We multiplied both costs by the number of PFOA/PFOS-attributable cases to identify total direct and indirect costs.

Results

Risk-of-bias tools yielded consistent evaluations of the quality of articles used as sources for ERRs across the two reviewers. Among the systematic reviews, the Steenland et al. review of associations with LBW was identified as having low risk of bias except for its reliance on a single source for studies (PubMed), failure to identify whether a single author or multiple authors assessed studies for inclusion and extracted data, and lack of risk-of-bias analysis. The Liu et al. review of childhood obesity and Kim et al. review of hypothyroidism were both identified to have low risk across all four domains by both authors. The Bartell et al. meta-analysis was based on data from articles identified in a prior review (Steenland and Winquist 2021) that also relied exclusively on PubMed, failed to identify who reviewed studies for inclusion and extracted data, and lacked a risk-of-bias analysis. Also, their calculations were based on data from only four studies of kidney cancer and two studies of testicular cancer. All four reviews appropriately considered the relevance of the identified research studies to the questions being considered and avoided emphasis on statistical significance, while two of the systematic reviews (Steenland and Bartell) were identified as probably (vs. conclusively) having addressed all concerns in the four domains. The overall risk of bias was identified as low for all four systematic reviews used (Table S15).

Both reviewers evaluated studies as definitely low risk of bias across all domains for subclinical hypothyroidism, T2D in females, childhood obesity, endometriosis, PCOS, and female breast cancer. The Meng et al. study of LBW, Zhang et al. study of GDM, and Impinem et al. study of pneumonia in children were judged by both reviewers to have potential live birth bias as a threat to internal validity, reducing the corresponding domain's assessment to probably low risk of bias. Both reviewers identified the Whitworth et al. study of couple infertility as potentially having conception bias, yielding a probably low risk of bias for internal validity, as well. One reviewer (LK) noted that the Liu et al. study of adult obesity used data collected as part of a randomized controlled trial, meaning the results may not be generalizable, and therefore evaluated the study as probably low risk of bias for internal validity. All of the evaluated studies at minimum had probably low risk of bias across all criteria, as evaluated by both reviewers (Table S16).

We identified PFOA-attributable disease costs in the US in 2018 of $5.52 billion across the five primary disease endpoints based on meta-analytic ERRs. This estimate represented the lower bound of possible costs, with our sensitivity analyses revealing as much as $62.6 billion in overall costs of long-chain PFAS exposure. Attributable fractions for PFAS of disease burden ranged from 0.08% for testicular cancer due to PFOA to 30.7% for LBW due to PFOS (Table 4).

The largest economic contributor to the main estimate of disease costs attributable to PFAS was childhood obesity ($2.65 billion). For childhood obesity, we also modeled a total lifetime direct medical cost of $4.56 billion due to PFOS exposure, which represents the incremental lifetime medical costs of a child becoming obese at age 10 relative to a child with normal BMI. Hypothyroidism in females contributed $1.26 billion in annual cost. This is a composite value of $42.1 million in direct costs of new cases of hypothyroidism and $1.22 billion in indirect costs as DALY lost over 10 years. The total PFOS-attributable cost for hypothyroidism in the sensitivity analysis was $5.18 billion. PFOA-attributable kidney and testicular cancer contributed a total of $4.88 million in direct costs and $187 million in indirect costs as DALY lost over 10 years. LBW due to PFOA exposure added $1.42 billion in healthcare expenditures annually; this estimate was a composite of $305 million due to direct costs of hospitalization associated with a LBW newborn and $1.11 billion attributable to lost IQ points associated with LBW. The cost estimates for PFOS exposure were substantially higher: $2.94 billion due to hospitalization costs and $10.7 billion due to lost IQ.

The highest costs we identified in both the main and sensitivity analyses were PFOS-attributable lifetime costs related to adult obesity, totaling $17.0 billion dollars annually. We estimated the PFOS-associated 15-year direct medical cost of obesity in newly obese 35-year olds as $3.21 billion, with $13.8 billion in QALY lost over the same 15 years. Other metabolic outcomes included T2D, for which the lifetime cost of PFOA-attributable annual incident cases in women was $140 million, and GDM, for which the low estimate totaled $414 million in annual costs due to PFOA exposure: $73.3 million in direct medical costs and $341 million in indirect costs of lost productivity secondary to adverse birth effects of GDM.

Women's gynecologic and reproductive health outcomes were also major contributors to the total calculated for the sensitivity analysis. Annual incident PFOA-attributable cases of endometriosis accounted for \$397 million to \$10.2 billion in total costs, with \$21.1 to \$547 million due to direct medical costs over 10 years and \$376 million to \$9.76 billion due to DALY lost over the same 10 years. The annual cost of initial evaluation of PCOS and treatment of associated menstrual dysfunction, infertility, T2D, and hirsutism generated at least \$10.5 million PFOA-attributable cost estimate, with a higher estimate of \$10.9 million. We estimated the cost of PFOA-attributable cases of couples seeking ART per annum to be at minimum \$37.6 million (\$1.66 billion upper bound) based on the price of a single embryo transfer ART cycle and the increase in medical costs associated with increased multiparity because of ART. For breast cancer due to PFOA exposure, we estimated \$159 million in direct medical cost of utilization of healthcare services within the first 6 months of a new breast cancer diagnosis and \$3.92 billion in DALY lost over 10 years. Finally, we estimated PFOA-attributable pneumonia in children < 3 years of age to cost the US medical system \$1.49 to \$22.5 billion annually due to treatment costs and indirect costs of parental absenteeism.

Discussion

PFAS contribute substantially to disease and disability in the US, with at least \$5.52 billion and as much as \$62.6 billion in associated economic costs. Our study builds on prior papers that have examined the disease burden and costs associated with PFAS exposure by incorporating 13 health outcomes for which evidence is strongest and constructing a range of models to estimate disease burden and economic costs. The findings suggest that the cost of remediation and of substituting PFAS with safer alternatives in consumer products may well be justified by the large economic costs of adverse health outcomes associated with PFAS exposure.

These estimates are highly conservative for multiple reasons. We did not include outcomes reported by the C8 Science Panel that were not confirmed in general population studies, as those associations were identified in a highly exposed population and our focus was on estimating the disease burden and economic costs due to routine exposure. We also did not include endpoints for which not enough consistent evidence has accumulated, such as prematurity, attention-deficit hyperactivity disorder, and lowered IQ in children resulting from prenatal exposure, and prostate cancer in adult men (Kahn et al. 2020). We based our minimum estimate on the costs associated with a single PFAS (PFOA) for each exposure-disease association and did not aggregate costs across multiple members of the PFAS class, when evidence suggests additivity and synergy in this class of > 4700 chemicals (Chohan et al. 2020). We quantified disease burden for only those associations with strongest scientific evidence for probable causation. We aggregated published costs for each of the diseases considered, but our calculations do not capture the real and substantial social costs such as pain and suffering to patients with PFAS-attributable conditions and effects on their loved ones (Cordner et al. 2021).

Our approach has several limitations. Our analysis relies on previously conducted studies to provide ERRs between PFOA/PFOS exposure and the outcomes of interest. These studies may not be generalizable to the current US population due to recent shifts away from the use of PFOA and PFOS in manufacturing; indeed, median serum levels of PFOA and PFOS in the US have declined substantially from 2007–2008 to 2017–2018, although production of—and consequent human exposure to—replacement PFAS, such as GenX, which are at least as toxic as PFOA (United States Environmental Protection Agency 2021), have increased. Despite the vast literature that exists on the endocrine-disrupting effects of PFAS, there have yet to be large cohort studies to evaluate the longitudinal effects of PFAS exposure in humans and decades of epidemiologic data are required before causation may be acknowledged and attributable disease burden calculated with more certainty. However, the risk-of-bias assessments yielded probably to definitely low risk of bias, and the stakes of inaction are high enough to justify action. It is also important to note that there is likely an overlap in some of the indirect costs modeled in our analysis due to high rates of comorbidity of endocrinopathies, e.g., the indirect lifetime costs of adult obesity may overlap with the costs of T2D.

Despite the limitations of our analysis, our models provide an approximation of the scope of the disease burden and associated costs attributable to exposure to these ubiquitous chemicals. As more research investigates the endocrine-disrupting effects of other chemicals in the PFAS class currently prevalent in manufacturing processes, it is likely that the PFAS-attributable disease burden and associated costs will continue to increase, further strengthening the case for regulation of the entire class of chemicals. Further action is urgently needed to limit these exposures from a health equity perspective, as exposure to these chemicals is not distributed equally throughout the US population and there are subsets who bear more of a burden, e.g., those who live near airports, military installations, and industrial plants (Attina et al. 2019).

Conclusion

The present study identifies at least \$5.52 billion in annual disease burden and associated social costs of current annual exposure to long-chain PFAS with our sensitivity analyses revealing as much as \$62.6 billion. Regulatory action to limit ongoing PFAS use and remediate contaminated water supplies may produce substantial economic benefits.

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Author Contributions LT conceptualized the study with VO. VO performed disease burden modeling and calculated economic costs. VO wrote the first draft, and LGK and LT provided input and final editorial oversight.

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Data Availability Data relating to this publication will be provided upon reasonable request.

Declarations

Conflict of interest LT acknowledges honoraria from Houghton Mifflin Harcourt, Audible, Paidos, and Kobunsha; travel support from the Endocrine Society, WHO, UNEP, Japan Environment and Health Ministries, and the American Academy of Pediatrics; as well as scientific advisory board activities for Beautycounter, IS-Global, and Footprint. All other authors declare they have nothing to disclose.

References

Aberger M, Wilson B, Holzbeierlein JM, Griebling TL, Nangia AK (2014) Testicular self-examination and testicular cancer: a cost-utility analysis. Cancer Med 3:1629–1634

Andrews DQ, Naidenko OV (2020) Population-wide exposure to per- and polyfluoroalkyl substances from drinking water in the United States. Environ Sci Technol Lett 7:931–936

Arbuckle TE, Kubwabo C, Walker M, Davis K, Lalonde K, Kosarac I et al (2013) Umbilical cord blood levels of perfluoroalkyl acids and polybrominated flame retardants. Int J Hyg Environ Health 216:184–194

Attina TM, Hauser R, Sathyanarayana S, Hunt PA, Bourguignon JP, Myers JP et al (2016) Exposure to endocrine-disrupting chemicals in the USA: a population-based disease burden and cost analysis. Lancet Diabetes Endocrinol 4:996–1003

Attina TM, Malits J, Naidu M, Trasande L (2019) Racial/ethnic disparities in disease burden and costs related to exposure to endocrine-disrupting chemicals in the United States: an exploratory analysis. J Clin Epidemiol 108:34–43

Azziz R, Woods KS, Reyna R, Key TJ, Knochenhauer ES, Yildiz BO (2004) The prevalence and features of the polycystic ovary syndrome in an unselected population. J Clin Endocrinol Metab 89:2745–2749

Azziz R, Marin C, Hoq L, Badamgarav E, Song P (2005) Health care-related economic burden of the polycystic ovary syndrome during the reproductive life span. J Clin Endocrinol Metab 90:4650–4658

Bach CC, Bech BH, Brix N, Nohr EA, Bonde JP, Henriksen TB (2015) Perfluoroalkyl and polyfluoroalkyl substances and human fetal growth: a systematic review. Crit Rev Toxicol 45:53–67

Bach CC, Bech BH, Nohr EA, Olsen J, Matthiesen NB, Bonefeld-Jorgensen EC et al (2016a) Perfluoroalkyl acids in maternal serum and indices of fetal growth: the Aarhus Birth Cohort. Environ Health Perspect 124:848–854

Bach CC, Vested A, Jorgensen KT, Bonde JP, Henriksen TB, Toft G (2016b) Perfluoroalkyl and polyfluoroalkyl substances and measures of human fertility: a systematic review. Crit Rev Toxicol 46:735–755

Bartell SM, Vieira VM (2021) Critical review on pfoa, kidney cancer, and testicular cancer. J Air Waste Manag Assoc 71:663–679

Bartell SM, Calafat AM, Lyu C, Kato K, Ryan PB, Steenland K (2010) Rate of decline in serum PFOA concentrations after granular activated carbon filtration at two public water systems in Ohio and West Virginia. Environ Health Perspect 118:222–228

Birnbaum LS (2018) Director, National institute of environmental health sciences and national toxicology program, national institutes of health, department of health and human services. Testimony on The Federal Role in the Toxic PFAS Crisis before the Senate Committee on Homeland Security and Governmental Affairs, Subcommittee on Federal Spending. Available at http://www.govinfo.gov

Boivin J, Bunting L, Collins JA, Nygren KG (2007) International estimates of infertility prevalence and treatment-seeking: potential need and demand for infertility medical care. Hum Reprod 22:1506–1512

Buck Louis GM, Peterson CM, Chen Z, Hediger ML, Croughan MS, Sundaram R et al (2012) Perfluorochemicals and endometriosis: the ENDO Study. Epidemiology 23:799–805

C8 Science Panel. C8 probable link reports. http://www.c8sciencepanel.org/prob_link.html

Calafat AM, Wong LY, Kuklenyik Z, Reidy JA, Needham LL (2007) Polyfluoroalkyl chemicals in the U.S. population: data from the National Health and Nutrition Examination Survey (NHANES) 2003–2004 and comparisons with NHANES 1999–2000. Environ Health Perspect 115:1596–1602

Cawley J, Meyerhoefer C (2012) The medical care costs of obesity: an instrumental variables approach. J Health Econ 31:219–230

CDC (2016) NHANES 2013–2014, laboratory methods and procedure manual. https://wwwn.cdc.gov/nchs/nhanes/continuousnhanes/manuals.aspx?BeginYear=2013

CDC (2018) Infertility. https://www.cdc.gov/nchs/fastats/infertility.htm

CDC (2020) National diabetes statistics report, 2020: estimates of diabetes and its burden in the United States. Center for Disease Control, Atlanta

CDC (2022a) Tenatus. https://www.Cdc.Gov/tetanus/about/index.Html#:~:Text=today%2c%20tetanus%20is%20uncommon%

20in,all%20the%20recommended%20tetanus%20vaccinations. Accessed 3 Mar 2022a

CDC (2022b) Diphtheria. https://www.Cdc.Gov/diphtheria/about/index.Html. Accessed 3 Mar 2022b

CDC/NCHS (2014) Birth data files. https://www.cdc.gov/nchs/data_access/vitalstatsonline.htm

CDC/NCHS (2018) Birth data files. https://www.cdc.gov/nchs/nvss/birth_methods.htm

Chohan A, Petaway H, Rivera-Diaz V, Day A, Colaianni O, Keramati M (2020) Per and polyfluoroalkyl substances scientific literature review: water exposure, impact on human health, and implications for regulatory reform. Rev Environ Health. https://doi.org/10.1515/reveh-2020-0049

Cordner A, Goldenman G, Birnbaum LS, Brown P, Miller MF, Mueller R et al (2021) The true cost of pfas and the benefits of acting now. Environ Sci Technol 55:9630–9633

Crawford S, Boulet SL, Mneimneh AS, Perkins KM, Jamieson DJ, Zhang Y et al (2016) Costs of achieving live birth from assisted reproductive technology: a comparison of sequential single and double embryo transfer approaches. Fertil Steril 105:444–450

Crone BC, Speth TF, Wahman DG, Smith SJ, Abulikemu G, Kleiner EJ et al (2019) Occurrence of per- and polyfluoroalkyl substances (PFAS) in source water and their treatment in drinking water. Crit Rev Environ Sci Technol 49:2359–2396

Ding T (2017) Epidemiological investigation and economic analysis of polycystic ovary syndrome (PCOS) for women in the UK. University College London, London

Eick SM, Goin DE, Chartres N, Lam J, Woodruff TJ (2020) Assessing risk of bias in human environmental epidemiology studies using three tools: different conclusions from different tools. Syst Rev 9:249

Environmental Protection Agency (2016) Drinking water health advisories for PFOA and PFOS. https://www.epa.gov/ground-water-and-drinking-water/drinking-water-health-advisories-pfoa-and-pfos 2020

Finkelstein EA, Graham WC, Malhotra R (2014) Lifetime direct medical costs of childhood obesity. Pediatrics 133:854–862

Fryar CD, Carroll MD, Ogden CL (2016) Prevalence of overweight, obesity, and extreme obesity among adults aged 20 and over: United states, 1960–1962 through 2013–2014. Center for Disease Control, Atlanta

Fuldeore MJ, Soliman AM (2017) Prevalence and symptomatic burden of diagnosed endometriosis in the United States: national estimates from a cross-sectional survey of 59,411 women. Gynecol Obstet Investig 82:453–461

Fuldeore M, Yang H, Du EX, Soliman AM, Wu EQ, Winkel C (2015) Healthcare utilization and costs in women diagnosed with endometriosis before and after diagnosis: a longitudinal analysis of claims databases. Fertil Steril 103:163–171

Garmendia Madariaga A, Santos Palacios S, Guillen-Grima F, Galofre JC (2014) The incidence and prevalence of thyroid dysfunction in europe: a meta-analysis. J Clin Endocrinol Metab 99:923–931

Goldenman G, Fernandes M, Holland M, Tugran T, Nordin A, Schoumacher C et al (2019) The cost of inaction: a socioeconomic analysis of environmental and health impacts linked to exposure to PFAS. Nordic Council of Ministers, Copenhagen

Gould E (2009) Childhood lead poisoning: conservative estimates of the social and economic benefits of lead hazard control. Environ Health Perspect 117:1162–1167

Grandjean P, Clapp R (2015) Perfluorinated alkyl substances: emerging insights into health risks. New Solut 25:147–163

Grandjean P, Andersen E, Budtz-Jorgensen E, Nielsen F, Molbak K, Weihe P (2012) Serum vaccine antibody concentrations in children exposed to perfluorinated compounds. JAMA 307:391–397

Hales CM, Carroll MD, Fryar CD, Ogden CL (2017) Prevalence of obesity among adults and youth: United States, 2015–2016. NCHS Data Brief 288:1–8

Hales CM, Carroll MD, Fryar CD, Ogden CL (2020) Prevalence of obesity and severe obesity among adults: United states, 2017–2018. NCHS Data Brief 360:1–8

Hepp Z, Lage MJ, Espaillat R, Gossain VV (2021) The direct and indirect economic burden of hypothyroidism in the United States: a retrospective claims database study. J Med Econ 24:440–446

Hill A (1965) The environment and disease: association or causation? Proc R Soc Med 58:295–300

Hollowell JG, Staehling NW, Flanders WD, Hannon WH, Gunter EW, Spencer CA et al (2002) Serum TSH, T(4), and thyroid antibodies in the United States population (1988 to 1994): National Health and Nutrition Examination Survey (NHANES III). J Clin Endocrinol Metab 87:489–499

Holzer J, Midasch O, Rauchfuss K, Kraft M, Reupert R, Angerer J et al (2008) Biomonitoring of perfluorinated compounds in children and adults exposed to perfluorooctanoate-contaminated drinking water. Environ Health Perspect 116:651–657

Hoylman B (2020) An act to amend the environmental conservation law, in relation to the use of perfluoroalkyl and polyfluoroalkyl substances in food packaging. Available at https://www.nysenate.gov/legislation/bills/2019/S8817

Impinen A, Longnecker MP, Nygaard UC, London SJ, Ferguson KK, Haug LS et al (2019) Maternal levels of perfluoroalkyl substances (PFASS) during pregnancy and childhood allergy and asthma related outcomes and infections in the norwegian mother and child (MoBa) cohort. Environ Int 124:462–472

Institute of Medicine (1981) Costs of environment-related health effects: a plan for continuing study. The National Academies Press, Washington (DC)

Jain S, Williams DJ, Arnold SR, Ampofo K, Bramley AM, Reed C et al (2015) Community-acquired pneumonia requiring hospitalization among U.S. Children. N Engl J Med 372:835–845

Johnson PI, Sutton P, Atchley DS, Koustas E, Lam J, Sen S et al (2014) The navigation guide—evidence-based medicine meets environmental health: systematic review of human evidence for PFOA effects on fetal growth. Environ Health Perspect 122:1028–1039

Kahn LG, Philippat C, Nakayama SF, Slama R, Trasande L (2020) Endocrine-disrupting chemicals: implications for human health. Lancet Diabetes Endocrinol 8:703–718

Kahn LG, Harley KG, Siegel EL, Zhu Y, Factor-Litvak P, Porucznik CA et al (2021) Persistent organic pollutants and couple fecundability: a systematic review. Hum Reprod Update 27:339–366

Kim MJ, Moon S, Oh BC, Jung D, Ji K, Choi K et al (2018) Association between perfluoroalkyl substances exposure and thyroid function in adults: a meta-analysis. PLoS ONE 13:e0197244

Knol MJ, Le Cessie S, Algra A, Vandenbroucke JP, Groenwold RH (2012) Overestimation of risk ratios by odds ratios in trials and cohort studies: alternatives to logistic regression. CMAJ 184:895–899

Kormos CE, Wilkinson AJ, Davey CJ, Cunningham AJ (2014) Low birth weight and intelligence in adolescence and early adulthood: a meta-analysis. J Public Health (oxf) 36:213–224

Koustas E, Lam J, Sutton P, Johnson PI, Atchley DS, Sen S et al (2014) The navigation guide—evidence-based medicine meets environmental health: systematic review of nonhuman evidence for PFOA effects on fetal growth. Environ Health Perspect 122:1015–1027

Kowlessar NM, Jiang J, Steiner C (2011) Hospital stays for newborns. Agency for Healthcare Research and Quality, Rockville

Lam J, Koustas E, Sutton P, Johnson PI, Atchley DS, Sen S et al (2014) The navigation guide—evidence-based medicine meets environmental health: integration of animal and human evidence for PFOA effects on fetal growth. Environ Health Perspect 122:1040–1051

Lamerato L, Havstad S, Gandhi S, Jones D, Nathanson D (2006) Economic burden associated with breast cancer recurrence: findings from a retrospective analysis of health system data. Cancer 106:1875–1882

Lauritzen HB, Larose TL, Oien T, Sandanger TM, Odland JO, van de Bor M et al (2018) Prenatal exposure to persistent organic pollutants and child overweight/obesity at 5-year follow-up: a prospective cohort study. Environ Health 17:9

Levin ML (1953) The occurrence of lung cancer in man. Acta Unio Int Contra Cancrum 9:531–541

Liu G, Dhana K, Furtado JD, Rood J, Zong G, Liang L et al (2018a) Perfluoroalkyl substances and changes in body weight and resting metabolic rate in response to weight-loss diets: a prospective study. PLoS Med 15:e1002502

Liu P, Yang F, Wang Y, Yuan Z (2018b) Perfluorooctanoic acid (PFOA) exposure in early life increases risk of childhood adiposity: a meta-analysis of prospective cohort studies. Int J Environ Res Public Health. https://doi.org/10.3390/ijerph15102070

Malits J, Blustein J, Trasande L, Attina TM (2018) Perfluorooctanoic acid and low birth weight: estimates of us attributable burden and economic costs from 2003 through 2014. Int J Hyg Environ Health 221:269–275

Max WB, Rice DP, Sung H, Michel M (2004) Valuing human life: estimating the present value of lifetime earnings, 2000. Available at https://escholarship.org/uc/item/82d0550k

Meng Q, Inoue K, Ritz B, Olsen J, Liew Z (2018) Prenatal exposure to perfluoroalkyl substances and birth outcomes; an updated analysis from the Danish National Birth Cohort. Int J Environ Rese Public Health 15(9):1832

Missmer SA, Hankinson SE, Spiegelman D, Barbieri RL, Marshall LM, Hunter DJ (2004) Incidence of laparoscopically confirmed endometriosis by demographic, anthropometric, and lifestyle factors. Am J Epidemiol 160:784–796

Muennig P, Lubetkin E, Jia H, Franks P (2006) Gender and the burden of disease attributable to obesity. Am J Public Health 96:1662–1668

Neumann PJ, Cohen JT, Weinstein MC (2014) Updating cost-effectiveness—the curious resilience of the $50,000-per-QALY Threshold. N Engl J Med 371:796–797

Office of Health Assessment and Translation (OHAT) (2022) Handbook for conducting a literature-based health assessment using ohat approach for systematic review and evidence integration. National Institute of Environmental Health Sciences; 2019. https://ntp.Niehs.Nih.Gov/ntp/ohat/pubs/handbookmarch2019_508.Pdf

Olsen GW, Mair DC, Reagen WK, Ellefson ME, Ehresman DJ, Butenhoff JL et al (2007) Preliminary evidence of a decline in perfluorooctanesulfonate (PFOS) and perfluorooctanoate (PFOA) concentrations in American Red Cross blood donors. Chemosphere 68:105–111

Pelch KE, Reade A, Wolffe TAM, Kwiatkowski CF (2019) PFAS health effects database: protocol for a systematic evidence map. Environ Int 130:104851

Peterson C, Grosse SD, Li R, Sharma AJ, Razzaghi H, Herman WH et al (2015) Preventable health and cost burden of adverse birth outcomes associated with pregestational diabetes in the United States. Am J Obstet Gynecol 212(74):e71-79

Rappazzo KM, Coffman E, Hines EP (2017) Exposure to perfluorinated alkyl substances and health outcomes in children: a systematic review of the epidemiologic literature. Int J Environ Res Public Health 14(7):691

Rogers RD, Reh CM, Breysse P (2021) Advancing per- and polyfluoroalkyl substances (PFAS) research: an overview of ATSDR and NCEH activities and recommendations. J Expo Sci Environ Epidemiol 31:961–971

Schrenk D, Bignami M, Bodin L, Chipman JK, Del Mazo J, Grasl-Kraupp B et al (2020) Risk to human health related to the presence of perfluoroalkyl substances in food. EFSA J Eur Food Saf Auth 18:e06223

Shih YT, Xu Y, Chien CR, Kim B, Shen Y, Li L et al (2019) Rising economic burden of renal cell carcinoma among elderly patients in the USA: part II—an updated analysis of SEER-Medicare Data. Pharmacoeconomics 37:1495–1507

Stahlman S, Fan M (2019) Female infertility, active component service women, U.S. Armed forces, 2013–2018. MSMR 26:20–27

Steenland K, Winquist A (2021) PFAS and cancer, a scoping review of the epidemiologic evidence. Environ Res 194:110690

Steenland K, Barry V, Savitz D (2018) Serum perfluorooctanoic acid and birthweight: an updated meta-analysis with bias analysis. Epidemiology 29:765–776

Steenland K, Fletcher T, Stein CR, Bartell SM, Darrow L, Lopez-Espinosa MJ et al (2020) Review: evolution of evidence on PFOA and health following the assessments of the C8 Science Panel. Environ Int 145:106125

Sun Q, Zong G, Valvi D, Nielsen F, Coull B, Grandjean P (2018) Plasma concentrations of perfluoroalkyl substances and risk of type 2 diabetes: a prospective investigation among U.S. women. Environ Health Perspect 126:037001

Surveillance Research Program NCI (2021) Surveillance, epidemiology, and end results program. Seer*explorer: an interactive website for seer cancer statistics. https://surveillance.cancer.gov/

Tong S, Amand C, Kieffer A, Kyaw MH (2018) Trends in healthcare utilization and costs associated with pneumonia in the United States during 2008–2014. BMC Health Serv Res 18:715

United States Environmental Protection Agency (2021) Human health toxicity assessments for GenX chemicals. https://www.epa.gov/chemical-research/human-health-toxicity-assessments-genx-chemicals. Accessed 2 Nov 2021

US Bureau of Labor Statistics (2020) CPI for all urban consumers (CPI-U). https://data.bls.gov/cgi-bin/surveymost?cu

U.S. Bureau of Labor Statistics (2021) Median usual weekly earnings of full-time wage and salary workers by sex, quarterly averages, seasonally adjusted. https://www.bls.gov/news.release/wkyeng.t01.htm

US Cancer Statistics Working Group (2020) U.S. Cancer statistics data visualizations tool, based on 2019 submission data (1999–2017). https://www.cdc.gov/cancer/dataviz

USC Bureau (2020) U.S. and world population clock. https://www.census.gov/popclock/2020

Vagi SJ, Azziz-Baumgartner E, Sjodin A, Calafat AM, Dumesic D, Gonzalez L et al (2014) Exploring the potential association between brominated diphenyl ethers, polychlorinated biphenyls, organochlorine pesticides, perfluorinated compounds, phthalates, and bisphenol a in polycystic ovary syndrome: a case–control study. BMC Endocr Disord 14:86

Wan MLY, Co VA, El-Nezami H (2021) Endocrine disrupting chemicals and breast cancer: a systematic review of epidemiological studies. Crit Rev Food Sci Nutr. https://doi.org/10.1080/10408398.2021.1903382

Waterfield G, Rogers M, Grandjean P, Auffhammer M, Sunding D (2020) Reducing exposure to high levels of perfluorinated compounds in drinking water improves reproductive outcomes: evidence from an intervention in Minnesota. Environ Health 19:42

Wen LL, Lin LY, Su TC, Chen PC, Lin CY (2013) Association between serum perfluorinated chemicals and thyroid function in U.S. adults: the national health and nutrition examination survey 2007–2010. J Clin Endocrinol Metab 98:E1456-1464

Whiting P, Savović J, Higgins JPT, Caldwell DM, Reeves BC, Shea B et al (2016) Robis: a new tool to assess risk of bias in systematic reviews was developed. J Clin Epidemiol 69:225–234

Whitworth KW, Haug LS, Baird DD, Becher G, Hoppin JA, Skjaerven R et al (2012) Perfluorinated compounds and subfecundity in pregnant women. Epidemiology 23:257–263

Wielsoe M, Kern P, Bonefeld-Jorgensen EC (2017) Serum levels of environmental pollutants is a risk factor for breast cancer in inuit: a case control study. Environ Health 16:56

Williams S, Gousen S, DeFrances C (2018) National hospital care survey demonstration projects: pneumonia inpatient hospitalizations and emergency department visits (National Health Statistics Reports). U.S. Department of Health and Human Services, Washington, D.C.

Xu Y, Fletcher T, Pineda D, Lindh CH, Nilsson C, Glynn A et al (2020) Serum half-lives for short- and long-chain perfluoroalkyl acids after ceasing exposure from drinking water contaminated by firefighting foam. Environ Health Perspect 128:77004

Zhang C, Sundaram R, Maisog J, Calafat AM, Barr DB, Buck Louis GM (2015) A prospective study of prepregnancy serum concentrations of perfluorochemicals and the risk of gestational diabetes. Fertil Steril 103:184–189

Zhou X, Zhang P, Hoerger TJ (2013) Lifetime direct medical costs of treating type 2 diabetes and diabetic complications. Am J Prev Med 45:253–261

ENVIRONMENTAL PROTECTION AGENCY

40 CFR Parts 141 and 142

[EPA–HQ–OW–2022–0114; FRL 8543–02–OW]

RIN 2040–AG18

PFAS National Primary Drinking Water Regulation

AGENCY: Environmental Protection Agency (EPA).

ACTION: Final rule.

SUMMARY: In March 2023, the U.S. Environmental Protection Agency (EPA) proposed and requested comment on the National Primary Drinking Water Regulation (NPDWR) and health-based Maximum Contaminant Level Goals (MCLGs) for six per- and polyfluoroalkyl substances (PFAS): perfluorooctanoic acid (PFOA), perfluorooctane sulfonic acid (PFOS), perfluorohexane sulfonic acid (PFHxS), perfluorononanoic acid (PFNA), hexafluoropropylene oxide dimer acid (HFPO–DA, commonly known as GenX Chemicals), and perfluorobutane sulfonic acid (PFBS). After consideration of public comment and consistent with the provisions set forth under the Safe Drinking Water Act (SDWA), the EPA is finalizing NPDWRs for these six PFAS. Through this action, the EPA is finalizing MCLGs for PFOA and PFOS at zero. Considering feasibility, the EPA is promulgating individual Maximum Contaminant Levels (MCLs) for PFOA and PFOS at 4.0 nanograms per liter (ng/L) or parts per trillion (ppt). The EPA is also finalizing individual MCLGs and is promulgating individual MCLs for PFHxS, PFNA, and HFPO–DA at 10 ng/L. In addition to the individual MCLs for PFHxS, PFNA, and HFPO–DA, in consideration of the known toxic effects, dose additive health concerns and occurrence and likely co-occurrence in drinking water of these three PFAS, as well as PFBS, the EPA is finalizing a Hazard Index (HI) of 1 (unitless) as the MCLG and MCL for any mixture containing two or more of PFHxS, PFNA, HFPO–DA, and PFBS. Once fully implemented, the EPA estimates that the rule will prevent thousands of deaths and reduce tens of thousands of serious PFAS-attributable illnesses.

DATES: This final rule is effective on June 25, 2024. The incorporation by reference of certain publications listed in the rule is approved by the Director of the **Federal Register** as of June 25, 2024.

ADDRESSES: The EPA has established a docket for this action under Docket ID No. EPA–HQ–OW–2022–0114. All documents in the docket are listed on the *http://www.regulations.gov* website. Although listed in the index, some information is not publicly available, *e.g.,* Confidential Business Information (CBI) or other information whose disclosure is restricted by statute. Certain other material, such as copyrighted material, is not placed on the internet and will be publicly available only in hard copy form. Publicly available docket materials are available electronically through *https://www.regulations.gov*.

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SUPPLEMENTARY INFORMATION:

Executive Summary

The Environmental Protection Agency (EPA) is issuing an adaptive and flexible National Primary Drinking Water Regulation (NPDWR) under the Safe Drinking Water Act (SDWA) to manage risks of per- and polyfluoroalkyl substances (PFAS) in drinking water. The EPA is establishing drinking water standards for six PFAS in this NPDWR to provide health protection against these individual and co-occurring PFAS in public water systems. The EPA's final rule represents data-driven drinking water standards that are based on the best available science and meet the requirements of SDWA. For the six PFAS, the EPA considered PFAS health effects information, evidence supporting dose-additive health concerns from co-occurring PFAS, as well as national and state data for the levels of multiple PFAS in finished drinking water. SDWA provides a framework for the EPA to regulate emerging contaminants of concern in drinking water. Under the statute, the EPA must act based on the "best available" science and information. Thus, the statute recognizes that the EPA may act in the face of imperfect information. It also provides a mechanism for the EPA to update standards as more science becomes available. For the PFAS covered by this rule, the EPA concluded that the state of the science and information has sufficiently advanced to the point to satisfy the statutory requirements and fulfill SDWA's purpose to protect public health by addressing contaminants in the nation's public water systems.

PFAS are a large class of thousands of organic chemicals that have unique physical and chemical properties. These compounds are designed to be stable and non-reactive because of the applications in which they are used: certain industrial and manufacturing processes; stain and water repellants in clothing, carpets, and other consumer products, as well as certain types of fire-fighting foams. PFAS tend to break down slowly and persist in the environment, and consequently, they can accumulate in the environment and the human body over time. Current scientific research and available evidence have shown the potential for harmful human health effects after being exposed to some PFAS. Although some PFAS have been phased out of use in the United States, they are still found in the environment and in humans based on biomonitoring data.

Drinking water is one of several ways people can be exposed to PFAS. The EPA's examination of drinking water data shows that different PFAS can often be found together and in varying combinations as mixtures. Additionally, decades of research demonstrates that exposure to mixtures of different chemicals can elicit dose-additive health effects: even if the individual chemicals are each present at levels considered "safe," the mixture may cause significant adverse health effects. The high likelihood for different PFAS to co-occur in drinking water; the additive health concerns when present in mixtures; the diversity and sheer number of PFAS; and their general presence and persistence in the environment and the human body are reflective of the environmental and public health challenges the American public faces with PFAS, which poses a particular threat for overburdened communities that experience disproportionate environmental impacts. The final NPDWR includes:

1. Individual Maximum Contaminant Levels (MCLs)
   a. Perfluorooctanoic acid (PFOA) MCL = 4.0 nanograms per liter or parts per trillion (ng/L or ppt)
   b. Perfluorooctane sulfonic acid (PFOS) MCL = 4.0 ng/L
   c. Perfluorohexane sulfonic acid (PFHxS) MCL = 10 ng/L
   d. Perfluorononanoic acid (PFNA) MCL = 10 ng/L
   e. Hexafluoropropylene oxide dimer acid (HFPO–DA) MCL = 10 ng/L

2. A Hazard Index MCL to account for dose-additive health effects for mixtures that could include two or more of four

PFAS (PFHxS, PFNA, HFPO–DA, and perfluorobutane sulfonic acid (PFBS)). The Hazard Index MCL defines when the combined levels of two or more of these four PFAS requires action. A PFAS mixture Hazard Index less than or equal to 1 (unitless) indicates a level at which no known or anticipated adverse effects on the health of persons occur and allows for an adequate margin of safety with respect to health risk associated with a mixture of PFAS in finished drinking water. A PFAS mixture Hazard Index greater than 1 (unitless) indicates an exceedance of the health protective level. To calculate the Hazard Index, a ratio is developed for each PFAS by dividing the measured level of the PFAS in drinking water by the level (in ng/L or ppt) below which adverse health effects are not likely to occur (*i.e.,* the Health Based Water Concentration or HBWC). The HBWCs for each PFAS in the Hazard Index are:

a. PFHxS = 10 ng/L or ppt
b. PFNA = 10 ng/L
c. HFPO–DA = 10 ng/L
d. PFBS = 2,000 ng/L

The individual PFAS ratios are then summed across the mixture to yield the Hazard Index MCL as follows:

$$HI\ MCL\ =\ \left(\frac{[HFPO-DA_{water} ng/L]}{[10\ ng/L]}\right) + \left(\frac{[PFBS_{water} ng/L]}{[2000\ ng/L]}\right) + \left(\frac{[PFNA_{water} ng/L]}{[10\ ng/L]}\right) + \left(\frac{[PFHxS_{water} ng/L]}{[10\ ng/L]}\right) = 1$$

Based on the administrative record for the final PFAS NPDWR and as discussed above, certain PFAS (including PFHxS, PFNA, HFPO–DA, and PFBS) have been shown to be toxicologically similar; *i.e.,* elicit the same or similar profile of adverse effects in several biological organs and systems (see USEPA, 2000a; USEPA, 2007; USEPA, 2024a; USEPA, USEPA, 2024c; and section IV.B of this preamble). Studies with PFAS and other classes of chemicals support the health-protective conclusion that chemicals that have similar observed adverse effects following individual exposure should be assumed to act in a dose-additive manner when in a mixture unless data demonstrate otherwise (USEPA, 2024a). Additionally, the record further supports that there is a substantial likelihood that PFBS, PFHxS, PFNA, and HFPO–DA co-occur as mixtures in drinking water at levels of public health concern (see USEPA, 2024b and sections VI.C and D of this preamble). Though the EPA is not promulgating an individual MCL or Maximum Contaminant Level Goal (MCLG) for PFBS at this time as it is for PFHxS, PFNA, and HFPO–DA (see section III.A of this preamble for specific discussion), based on these evaluations, the agency is establishing a Hazard Index MCL that addresses PFBS as part of mixtures where its co-occurrence with other PFAS (PFHxS, HFPO–DA, and/or PFNA) can affect health endpoints when present in these mixtures.

The individual and Hazard Index MCLs are independently applicable for compliance purposes.

Additionally, the EPA is finalizing important public ''right to know'' provisions of the EPA's SDWA regulations, specifically, public notification (PN) and Consumer Confidence Report (CCR) requirements. The changes under this rule will strengthen risk communication and education for the public when elevated levels of these PFAS are found. Finally, the EPA is finalizing monitoring and reporting requirements that enable public water systems (PWSs) and primacy agencies to implement and comply with the NPDWR.

Consistent with the timelines set out under SDWA, PWSs are required to conduct their initial monitoring by April 26, 2027, and to conduct PN and include PFAS information in the CCR. After carefully considering public comment, the EPA is extending the compliance deadline for all systems nationwide to meet the MCL to allow additional time for capital improvements. As such, PWSs are required to make any necessary capital improvements and comply with the PFAS MCLs by April 26, 2029.

As part of its *Health Risk Reduction and Cost Analysis* (HRRCA), the EPA evaluated quantifiable and nonquantifiable health risk reduction benefits and costs associated with the final NPDWR. At a two percent discount rate, the EPA estimates the quantifiable annual benefits of the final rule will be $1,549.40 million per year and the quantifiable costs of the rule will be $1,548.64 million per year. The EPA's quantified benefits are based on the agency's estimates that that there will be 29,858 fewer illnesses and 9,614 fewer deaths in the communities in the decades following actions to reduce PFAS levels in drinking water. While the modeled quantified net benefits are nearly at parity, under SDWA, the EPA must consider whether the costs of the rule are justified by the benefits based on all statutorily prescribed costs and benefits, not just the quantified costs and benefits (see SDWA 1412(b)(3)(c)(i)).

The EPA expects that the final rule will result in additional nonquantifiable costs, including costs with generally greater uncertainty, which the EPA has examined in quantified sensitivity analyses in the Economic Analysis for the final rule. First, the EPA had insufficient nationally representative data to precisely characterize occurrence of HFPO–DA, PFNA, and PFBS. In an effort to better consider and understand the costs associated with treatment of these regulated compounds at systems both with and without PFOA, PFOS and PFHxS occurrence in exceedance of the MCLs, the EPA performed a quantitative sensitivity analysis of the costs associated with Hazard Index and/or MCL exceedances resulting from HFPO–DA, PFNA, and PFBS. The EPA expects that the quantified national costs, which do not include HFPO–DA, PFNA, and PFBS treatment costs are marginally underestimated (on the order of 5 percent). Second, stakeholders have expressed concern to the EPA that a hazardous substance designation for certain PFAS may limit their disposal options for drinking water treatment residuals (*e.g.,* spent media, concentrated waste streams) and/or potentially increase costs. The EPA has conducted a sensitivity analysis and found that should all water systems use hazardous waste disposal options national costs would increase by 7 percent.

The EPA anticipates significant additional benefits that cannot be quantified, will result from avoided negative developmental, cardiovascular, liver, immune, endocrine, metabolic, reproductive, musculoskeletal, and carcinogenic effects as a result of reductions in the levels of the regulated PFAS and other co-removed contaminants. For example, elevated concentrations of both PFOA and PFOS negatively impact the immune and endocrine systems, impacts which the agency is unable to quantify at this time. As another example, the EPA assessed the developmental benefits associated with PFNA exposure reductions semi-quantitively in sensitivity analysis, and the analysis demonstrates significant additional benefits associated with reductions in PFNA. There are other nonquantifiable benefits for other PFNA health endpoints, and numerous endpoints for PFHxS, HFPO–DA, PFBS, and other PFAS that are anticipated to be removed as a result of the final NPDWR. Additionally, as a result of the ability for available treatment technologies to remove co-occurring contaminants, there are benefits not quantified for removal of co-occurring contaminants for this regulation (*e.g.,* certain pesticides, volatile organic compounds). Considering both quantifiable and nonquantifiable costs and benefits of the rule, the EPA is reaffirming the Administrator's determination at the time of proposal, that the quantifiable and nonquantifiable benefits of the final rule justify the quantifiable and nonquantifiable costs.

To help communities on the frontlines of PFAS contamination, the passage of the Infrastructure Investment and Jobs Act (IIJA), also referred to as the Bipartisan Infrastructure Law (BIL), invests billions of dollars over a 5-year period. BIL appropriates over $11.7 billion in the Drinking Water State Revolving Fund (DWSRF) General Supplemental; $4 billion to the DWSRF for Emerging Contaminants; and $5 billion in grants to the Emerging Contaminants in Small or Disadvantaged Communities. These funds will assist many disadvantaged communities, small systems, and others with the costs of installation of treatment when it might otherwise be cost-challenging.

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I. General Information

A. What are the EPA's final rule requirements?

The Safe Drinking Water Act (SDWA) provides a framework for the Environmental Protection Agency (EPA) to regulate emerging contaminants of concern in drinking water. Under the statute, the EPA may act based on the ''best available'' science and information. Thus, the statute recognizes that the EPA may act in the face of imperfect information and provides a mechanism for the EPA to update standards as more science becomes available. For the per- and polyfluoroalkyl substances (PFAS) covered by this rule, the EPA concluded that the state of the science and information has sufficiently advanced to the point to satisfy the statutory requirements and fulfill SDWA's purpose to protect public health by addressing contaminants in the nation's public water systems. In this final action, the EPA is finalizing the PFAS National Primary Drinking Water Regulation (NPDWR) that is based upon the best available peer-reviewed science. The final NPDWR for PFAS establishes Maximum Contaminant Level Goals (MCLGs) and enforceable Maximum Contaminant Levels (MCLs) for six PFAS compounds: perfluorooctanoic acid (PFOA), perfluorooctane sulfonic acid (PFOS), perfluorohexane sulfonic acid (PFHxS), perfluorononanoic acid (PFNA), hexafluoropropylene oxide dimer acid (HFPO–DA, commonly known as GenX Chemicals), and perfluorobutane sulfonic acid (PFBS). The final rule requirements and references to where additional discussion can be found on these topics are summarized here:

The EPA is finalizing MCLGs for PFOA and PFOS at zero (0) and enforceable MCLs for PFOA and PFOS at 4.0 ng/L (ng/L or ppt). Please see section IV of this preamble on the MCLG derivation for PFOA and PFOS. Additionally, please see section V of this preamble for discussion on the MCL for PFOA and PFOS.

The EPA is finalizing individual regulatory determinations to regulate PFHxS, PFNA, and HFPO–DA (commonly known as ''GenX Chemicals''). The EPA is deferring the individual regulatory determination to regulate PFBS in drinking water. Concurrent with the final determinations, the EPA is promulgating individual MCLGs and MCLs for PFHxS, PFNA, and HFPO–DA at 10 ng/L each.

Additionally, the EPA is finalizing a regulatory determination for mixtures of PFHxS, PFNA, HFPO–DA, and PFBS due to their substantial likelihood for co-occurrence and dose-additive health concerns when present as a mixture in drinking water. Concurrent with this final determination, the EPA is finalizing a Hazard Index (HI) of 1 as the MCLG and enforceable MCL to address mixtures of PFHxS, PFNA, HFPO–DA, and PFBS where they co-occur in drinking water. Please see section III of this preamble for discussion on the EPA's final regulatory determinations; section IV of this preamble for discussion on the MCLG derivation for these additional compounds; and section V of this preamble for a discussion on the final MCLs.

This action also lists feasible technologies for public water systems (PWSs) that can be used to comply with the MCLs. The EPA notes that systems are not required to use the listed technologies to meet the MCL; rather, the MCL is a numeric regulatory limit systems must meet that is developed while considering treatment feasibility and cost. Please see section X for additional discussion on feasible treatment technologies.

The EPA is finalizing SDWA Right-to-Know requirements for the final rule, including Consumer Confidence Report (CCR) and Public Notification (PN) requirements. Community water systems (CWSs) must prepare and deliver to its customers an annual CCR in accordance with 40 CFR part 141, subpart O. Under this rule, CWSs will be required to report detected PFAS in their CCRs and provide health effects language in the case of MCL violations. Additionally, under the final rule, MCL violations require Tier 2 public notification, or notification provided as soon as practicable but no later than 30 days after a system learns of the violation, as per 40 CFR 141.203. Additionally, monitoring and testing procedure violations require Tier 3 notification, or notice no later than one year after the system learns of the violation. Please see section IX of this preamble for additional discussion on SDWA Right-to-Know requirements.

Additionally, the EPA is finalizing monitoring and reporting requirements for PWSs to comply with the NPDWR. PWSs are required to sample each EP using a monitoring regime generally based on the EPA's Standard Monitoring Framework (SMF) for Synthetic Organic Contaminants (SOCs). As a part of these requirements, to establish baseline levels of regulated PFAS, water systems must complete initial monitoring within three years following rule promulgation and/or use results of recent, previously acquired monitoring to satisfy the initial monitoring requirements. Following initial monitoring, beginning three years following rule promulgation, to demonstrate that finished drinking water does not exceed the MCLs for regulated PFAS, PWSs will be required to conduct compliance monitoring for all regulated PFAS at a frequency specifically based on sample results. Compliance with the NPDWRs will be based on analytical results obtained at each sampling point. PWSs are required to report to primacy agencies the results of all initial and compliance monitoring to ensure compliance with the NPDWRs. Please see section VIII of this preamble for additional discussion on these requirements.

Finally, the EPA is exercising its authority under SDWA section 1412(b)(10) to implement a nationwide capital improvement extension to comply with the MCL. All systems must comply with the MCLs by April 26, 2029. All systems must comply with all other requirements of the NPDWR, including initial monitoring, by April 26, 2027. For additional discussion on extensions and exemptions, please see section XI.

B. Does this action apply to me?

Entities regulated by this action are CWSs and non-transient non-community water systems (NTNCWSs). A PWS, as defined in 40 CFR 141.2, provides water to the public for human consumption through pipes or ''other constructed conveyances, if such system has at least fifteen service connections or regularly serves an average of at least twenty-five individuals daily at least 60 days out of the year.'' A PWS is either a CWS or a non-community water system (NCWS). A CWS, as defined in § 141.2, is ''a public water system which serves at least fifteen service connections used by year-round residents or regularly serves at least twenty-five year-round residents.'' The definition in § 141.2 for a NTNCWS is ''a public water system that is not a [CWS] and that regularly serves at least 25 of the same persons over 6 months per year.'' The following table provides examples of the regulated entities under this rule:

Category	Examples of potentially affected entities
Public water systems	CWSs; NTNCWSs.
State and Tribal agencies	Agencies responsible for drinking water regulatory development and enforcement.

This table is not intended to be exhaustive, but rather provides a guide for readers regarding entities likely to be regulated by this action. This table includes the types of entities that the EPA is now aware could potentially be regulated by this action. To determine whether your entity is regulated by this action, this final rule should be carefully examined. If you have questions regarding the applicability of this action to a particular entity, consult the person listed in the **FOR FURTHER INFORMATION CONTACT** section.

All new systems that begin operation after, or systems that use a new source of water after, April 26, 2024, must demonstrate compliance with the MCLs within a period of time specified by the Primacy Agency. The EPA has defined in 40 CFR chapter I, subchapter D, part 141, § 141.2, a wholesale system as a PWS that supplies finished PWSs and a consecutive system as a PWS that buys or otherwise receives some or all its finished water from a wholesale system. In this action, the EPA reiterates that all CWS and NTNCWS must comply with this regulation. This includes consecutive CWS and NTNCWS systems; however, the requirements these consecutive systems must implement to comply with the regulation may be, and often are, much less extensive. For finished water that is provided through a system interconnection, the wholesale systems will be responsible for conducting the monitoring requirements at the entry point (EP) to the distribution system. The final regulation does not require that any monitoring be conducted at a system interconnection point. Where a violation does occur, the wholesale system must notify any consecutive systems of this violation and it is the responsibility of the consecutive system to provide PN to their customers pursuant to § 141.201(c)(1). In addition, wholesale systems must also provide information in Subpart O to consecutive systems for developing CCRs (§ 141.201(c)(1)). Consecutive systems are responsible for providing their customers with the reports (§ 141.153(a)).

II. Background

A. What are PFAS?

Per- and polyfluoroalkyl substances (PFAS) are a large class of thousands of synthetic chemicals that have been in use in the United States and around the world since the 1940s (USEPA, 2018a). The ability for PFAS to withstand heat and repel water and stains makes them useful in a wide variety of consumer, commercial, and industrial products, and in the manufacturing of other products and chemicals. This rule applies directly to six specific PFAS: perfluorooctanoic acid (PFOA), perfluorooctane sulfonic acid (PFOS), perfluorononanoic acid (PFNA), hexafluoropropylene oxide dimer acid (HFPO–DA, commonly known as GenX Chemicals), perfluorohexane sulfonic acid (PFHxS), and perfluorobutane sulfonic acid (PFBS). Due to their widespread use, physicochemical properties, and prolonged persistence, many PFAS co-occur in air, water, ice, and soil, and in organisms, such as humans and wildlife. Exposure to some PFAS can lead to bioaccumulation in tissues and blood of aquatic as well as terrestrial organisms, including humans (Domingo and Nadal, 2019; Fromme et al., 2009). Pregnant and lactating women, as well as infants and children, may be more sensitive to the harmful effects of certain PFAS, such as PFOA, PFOS, PFNA, and PFBS. For example, studies indicate that PFOA and PFOS exposure above certain levels may result in adverse health effects, including developmental effects to fetuses during pregnancy or to breast- or formula-fed infants, increased risk for certain cancers, and negative immunological effects, among others (USEPA, 2024c; USEPA, 2024d). It has been documented that exposure to other PFAS are associated with a range of adverse health effects (USEPA, 2021a; USEPA, 2021b; ATSDR, 2021; NASEM, 2022).

The Environmental Protection Agency (EPA) is aware that PFAS still enter the environment and there are viable pathways for human exposure. Most United States production of PFOA, PFOS, and PFNA, along with other long-chain PFAS, was phased out and then generally replaced by production of PFHxS, HFPO–DA, PFBS, and other PFAS. The EPA is also aware of ongoing use of PFOA, PFOS, PFNA, and other long-chain PFAS (USEPA, 2000b; ATSDR, 2021). Long-chain PFAS are typically defined as including perfluoroalkyl sulfonic acids containing ≥ 6 carbons, and perfluoroalkyl carboxylic acids with ≥7 carbons. While domestic production and import of PFOA has been phased out in the United States by the companies participating in the 2010/2015 PFOA Stewardship Program, small quantities of PFOA may be produced, imported, and used by companies not participating in the PFOA Stewardship Program (USEPA, 2021c). The EPA is also aware of ongoing use of PFAS available from existing stocks or newly introduced via imports (see USEPA, 2022a). Additionally, the environmental persistence of these chemicals and formation as degradation products from other compounds may contribute to their ongoing release in the environment (ATSDR, 2021).

The six PFAS in this rule and their relevant Chemical Abstract Service registry numbers (CASRNs) are:

- PFOA ($C_8F_{15}O_2^-$; CASRN: 45285–51–6)
- PFOS ($C_8F_{17}SO_3^-$; CASRN: 45298–90–6)
- PFHxS ($C_6F_{13}SO_3^-$; CASRN: 108427–53–8)
- PFNA ($C_9F_{17}O_2^-$; CASRN: 72007–68–2)
- HFPO–DA ($C_6F_{11}O_3^-$; CASRN: 122499–17–6)
- PFBS ($C_4F_9SO_3^-$; CASRN: 45187–15–3)

These PFAS may exist in multiple forms, such as isomers or associated salts, and each form may have a separate CAS registry number or no CASRN at all. Additionally, these compounds have various names under different classification systems. However, at environmentally relevant pHs, these PFAS are expected to dissociate in water to their anionic (negatively charged) forms. For instance, International Union of Pure and Applied Chemistry substance 2,3,3,3-tetrafluoro-2-(heptafluoropropoxy) propanoate (CASRN: 122499–17–6), also known as HFPO–DA, is an anionic molecule which has an ammonium salt (CASRN: 62037–80–3), a conjugate acid (CASRN: 13252–13–6), a potassium salt (CASRN: 67118–55–2), and an acyl fluoride precursor (CASRN: 2062–98–8), among other variations. At environmentally relevant pHs these all dissociate into the propanoate/anion form (CASRN: 122499–17–6). Each PFAS listed has multiple variants with differing chemical connectivity, but the same molecular composition (known as isomers). Commonly, the isomeric composition of PFAS is categorized as 'linear,' consisting of an unbranched alkyl chain, or 'branched,' encompassing a potentially diverse group of molecules including at least one, but potentially more, offshoots from the linear molecule. While broadly similar, isomeric molecules may have differences in chemical properties. This rule covers all salts, isomers and derivatives of the chemicals listed, including derivatives other than the anionic form which might be created or identified.

B. Human Health Effects

The publicly available landscape of human epidemiological and experimental animal-based exposure-effect data from repeat-dose studies across PFAS derive primarily from carboxylic and sulfonic acid species such as PFOA, PFOS, PFHxS, PFNA, HFPO–DA, and PFBS (ATSDR, 2021; USEPA, 2021a; USEPA, 2021b; USEPA, 2024c; USEPA, 2024d). Many other PFAS have some human health effects data available (Mahoney et al., 2022) and some PFAS, such as PFBS, HFPO–DA, PFNA, and PFHxS, have sufficient data that has allowed Federal agencies to publish toxicity assessments (USEPA, 2021a; USEPA, 2021b; USEPA, 2024c; USEPA, 2024d; ATSDR, 2021) and derive toxicity values (*e.g.,* a reference dose), which is an estimate of daily exposure to the human population (including sensitive populations) that is likely to be without an appreciable risk of deleterious effects during a lifetime). The adverse health effects associated with exposure to such PFAS include (but are not limited to): effects on the liver (*e.g.,* liver cell death), growth and development (*e.g.,* low birth weight), hormone levels, kidney, the immune system (reduced response to vaccines), lipid levels (*e.g.,* high cholesterol), the nervous system, and reproduction, as well as increased risk of certain types of cancer.

Exposure to PFAS may have disproportionate health effects on children. Adverse health effects relevant to children associated with exposure to some PFAS include developmental effects to fetuses during pregnancy or to breast-fed infants, cardiovascular effects, immune effects, endocrine effects, and reproductive effects. Additionally, PFAS are known to be transmitted to the fetus via the placenta and to the newborn, infant, and child via breast milk (USEPA, 2021a; USEPA, 2021b; USEPA, 2024c; USEPA, 2024d; ATSDR, 2021).

Please see sections III.B and IV of this rule for additional discussion on health considerations for the six PFAS the EPA is regulating in this document.

C. Statutory Authority

Section 1412(b)(1)(A) of SDWA requires the EPA to establish National Primary Drinking Water Regulations (NPDWRs) for a contaminant where the Administrator determines that the contaminant: (1) may have an adverse effect on the health of persons; (2) is known to occur or there is a substantial likelihood that the contaminant will occur in PWSs (public water systems) with a frequency and at levels of public health concern; and (3) in the sole judgment of the Administrator, regulation of such contaminant presents a meaningful opportunity for health risk reduction for persons served by PWSs.

D. Statutory Framework and PFAS Regulatory History

Section 1412(b)(1)(B)(i) of the Safe Drinking Water Act (SDWA) requires the EPA to publish a Contaminant Candidate List (CCL) every five years. The CCL is a list of contaminants that are known or anticipated to occur in PWSs, are not currently subject to any proposed or promulgated NPDWRs and may require regulation under the drinking water program. In some cases, developing the CCL may be the first step in evaluating drinking water contaminants. The EPA uses the CCL to identify priority contaminants for regulatory decision-making (*i.e.,* regulatory determinations), and for data collection. Publishing a CCL does not impose any requirements on PWSs. The EPA included PFOA and PFOS on the third and fourth CCLs published in 2009 (USEPA, 2009a) and 2016 (USEPA, 2016a). The EPA then included PFAS as a chemical group in its most recent list, the fifth CCL (CCL 5) (USEPA, 2022b). This group is inclusive of the PFAS the EPA is regulating through this action; however, the fifth CCL did not include PFOA and PFOS as they had already had final positive regulatory determinations completed for them in March 2021 (USEPA, 2021d).

The EPA collects data on the CCL contaminants to better understand their potential health effects and to determine the levels at which they occur in PWSs. SDWA 1412(b)(1)(B)(ii) requires that, every five years and after considering public comments on a ‘‘preliminary’’ regulatory determination, the EPA issues a determination to regulate or not regulate at least five contaminants on each CCL. In addition, section 1412(b)(1)(B)(ii)(III) authorizes the EPA to make a determination to regulate a contaminant not listed on the CCL at any time so long as the contaminant meets the three statutory criteria based on available public health information. SDWA 1412(b)(1)(B)(iii) requires that ‘‘each document setting forth the determination for a contaminant under clause (ii) shall be available for public comment at such time as the determination is published.’’ To implement these requirements, the EPA issues preliminary regulatory determinations subject to public comment and then issues a final regulatory determination after consideration of public comment. Section 1412(b)(1)(E) requires that the EPA propose an NPDWR no later than 24 months after a final determination to regulate. The statute also authorizes the EPA to issue a proposed rule concurrent with a preliminary determination to regulate. The EPA must then promulgate a final regulation within 18 months of the proposal (which may be extended by 9 additional months).

The EPA also implements a monitoring program for unregulated contaminants under SDWA 1445(a)(2) that requires the EPA to issue a list once every five years of priority unregulated contaminants to be monitored by PWSs. This monitoring is implemented through the Unregulated Contaminant Monitoring Rule (UCMR), which collects data from community water systems (CWSs) and non-transient community water systems (NTNCWSs) to better improve the EPA’s understanding of the frequency of unregulated contaminants of concern occurring in the nation’s drinking water systems and at what levels. The first four UCMRs collected data from a census of large water systems (serving more than 10,000 people) and from a statistically representative sample of small water systems (serving 10,000 or fewer people).

Between 2013–2015, water systems collected monitoring data for six PFAS (PFOA, PFOS, PFHxS, PFNA, PFBS, and perfluoroheptanoic acid (PFHpA)) as part of the third UCMR (UCMR 3) monitoring program. The fifth UCMR (UCMR 5), published December 2021, requires sample collection and analysis for 29 PFAS, including PFOA, PFOS, PFHxS, PFNA, HFPO–DA, and PFBS, to occur between January 2023 and December 2025 using drinking water analytical methods developed by the EPA. Section 2021 of America’s Water Infrastructure Act of 2018 (AWIA) (Pub. L. 115–270) amended SDWA and specifies that, subject to the availability of the EPA appropriations for such purpose and sufficient laboratory capacity, the EPA must require all public water systems (PWSs) serving between 3,300 and 10,000 people to monitor and ensure that a nationally representative sample of systems serving fewer than 3,300 people monitor for the contaminants in UCMR 5 and future UCMR cycles. All large water systems continue to be required to participate in the UCMR program. Section VI of this preamble provides additional discussion on PFAS occurrence. While the complete UCMR 5 dataset was not available to inform this rule and thus not a basis for informing the agency’s decisions for the final rule, the EPA acknowledges that the small subset of data released (7 percent of the total results that the EPA expects to receive) as of July 2023 confirms the EPA’s conclusions supported by the extensive amount of data utilized in its UCMR 3, state data, and modelling analyses. This final rule allows utilities and primacy agencies to use the UCMR 5 data to support implementation of monitoring requirements. Sections VI and VIII of this preamble further discusses these occurrence analyses as well as monitoring and compliance requirements, respectively.

After careful consideration of public comments, the EPA issued final regulatory determinations for contaminants on the fourth CCL (CCL 4) in March of 2021 (USEPA, 2021d) which included determinations to regulate two contaminants, PFOA and PFOS, in drinking water. The EPA found that PFOA and PFOS may have an adverse effect on the health of persons; that these contaminants are known to occur, or that there is a substantial likelihood that they will occur, in PWSs with a frequency and at levels that present a public health concern; and that regulation of PFOA and PFOS presents a meaningful opportunity for health risk reduction for persons served by PWSs. As discussed in the final Regulatory Determinations 4 Notice for CCL 4 contaminants (USEPA, 2021d) and the EPA's *PFAS Strategic Roadmap* (USEPA, 2022c), the agency has also evaluated additional PFAS chemicals for regulatory consideration as supported by the best available science. The agency finds that additional PFAS compounds also meet SDWA criteria for regulation. The EPA's regulatory determination for these additional PFAS is discussed in section III of this preamble.

Section 1412(b)(1)(E) provides that the Administrator ''may publish such proposed regulation concurrent with the determination to regulate.'' The EPA interprets this provision as allowing concurrent processing of a preliminary determination with a proposed rule, not a final determination (as urged by some commenters—see responses in section III of this preamble). Under this interpretation, section 1412(b)(1)(E) authorizes the EPA to issue a preliminary determination to regulate a contaminant and a proposed NPDWR addressing that contaminant concurrently and request public comment at the same time. This represents the only interpretation that accounts for the statutory language in context and is the only one that fulfills Congress's purpose of permitting the agency to adjust its stepwise processes where appropriate to avoid any unnecessary delay in regulating contaminants that meet the statutory criteria. To the extent the statute is ambiguous, the EPA's interpretation is the best interpretation of this provision for these same reasons. As a result, this rule contains both a final determination to regulate four PFAS contaminants (individually and/or as part of a PFAS mixture), and regulations for those contaminants as well as the two PFAS contaminants (PFOA and PFOS) for which the EPA had already issued a final Regulatory Determination. The EPA developed an MCLG and an NPDWR for six PFAS compounds pursuant to the requirements under section 1412(b)(1)(B) of SDWA. The final Maximum Contaminant Level Goals (MCLGs) and NPDWR are discussed in more detail in the following section.

E. Bipartisan Infrastructure Law

The passage of the Infrastructure Investment and Jobs Act (IIJA), often referred to as the Bipartisan Infrastructure Law or BIL, invests over $50 billion to improve drinking water, wastewater, and stormwater infrastructure—the single largest investment in water by the Federal Government. This historic investment specific to safe drinking water includes $11.7 billion in the Drinking Water State Revolving Fund (DWSRF) General Supplemental (referred to as BIL DWSRF General Supplemental); $4 billion to the Drinking Water SRF for Emerging Contaminants (referred to as BIL DWSRF EC); and $5 billion in grants for Emerging Contaminants in Small or Disadvantaged Communities (referred to as EC–SDC) from Federal fiscal years 2022 through 2026 (USEPA, 2023a). For the BIL DWSRF General Supplemental and BIL DWSRF EC, states must provide 49% and 100%, respectively, as additional subsidization in the form of principal forgiveness and/or grants. The EC–SDC grant has no cost-share requirement. Together, these funds will assist many disadvantaged communities, small systems, and others with the costs of addressing emerging contaminants, like PFAS, when it might otherwise be cost-challenging. This financial assistance can be used to address emerging contaminants in drinking water through actions such as technical assistance, certain water quality testing, operator and contractor training and equipment, and treatment upgrades and expansion. Investments in these areas which will allow communities additional funding to meet their obligations under this regulation and help ensure protection from PFAS contamination of drinking water. The Drinking Water SRF can be used by water systems to reduce the public health concerns around PFAS in their drinking water and is already being successfully utilized. Additionally, to support BIL implementation, the EPA is offering water technical assistance (WaterTA) to help communities identify water challenges and solutions, build capacity, and develop application materials to access water infrastructure funding (USEPA, 2023b). The EPA collaborates with states, Tribes, territories, community partners, and other stakeholders with the goal of more communities with applications for Federal funding, quality water infrastructure, and reliable water services.

F. EPA PFAS Strategic Roadmap

In October 2021, the EPA published the *PFAS Strategic Roadmap* (or *Roadmap*) that outlined the whole-of-agency approach to ''further the science and research, to restrict these dangerous chemicals from getting into the environment, and to immediately move to remediate the problem in communities across the country'' (USEPA, 2022c). The *Roadmap* offers timelines by which the EPA acts on key commitments the agency made toward addressing these contaminants in the environment, while continuing to safeguard public health. These include the EPA proposing to designate certain PFAS as Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) hazardous substances; issuing advance notice of proposed rulemakings on various PFAS under CERCLA; and issuing updated guidance on destroying and disposing of certain PFAS and PFAS-containing materials. Additionally, the EPA is issued a memorandum to states in December 2022 that provides direction on how to use the National Pollutant Discharge Elimination System (NPDES) program to protect against PFAS (USEPA, 2022d; USEPA, 2022e). The EPA also announced revisions to several Effluent Limitation Guidelines (ELGs) including, Organic Chemical, Plastic, Synthetic Fibers manufacturing, Metal Finishing & Electroplating, and Landfills to address PFAS discharge from these point source categories. These ELGs collectively will, if finalized, restrict and reduce PFAS discharges to waterways used as sources for drinking water. The EPA is taking numerous other actions to advance our ability to understand and effectively protect people from PFAS, such as the October 11, 2023, rule finalized under the Toxic Substances Control Act (TSCA) that will provide the EPA, its partners, and the public with a dataset of PFAS manufactured and used in the United States. The rule requires all manufacturers (including importers) of PFAS and PFAS-containing articles in any year since 2011 to report information to the extent known or reasonably ascertainable: chemical identity, uses, volumes made and processed, byproducts, environmental and health effects, worker exposure, and disposal to the EPA. With this final NPDWR, the EPA is delivering on another key goal in the *Roadmap* to ''establish a National Primary Drinking Water Regulation'' for PFAS. This rule will protect the American people directly from everyday PFAS exposures that might otherwise occur from PFAS-contaminated drinking water, complementing the many other actions in the *Roadmap* to protect public health and the environment from PFAS.

III. Final Regulatory Determinations for Additional PFAS

A. Agency Findings

As noted earlier, in 2021, the EPA made a determination to regulate two per- and polyfluoroalkyl substances—perfluorooctanoic acid (PFOA) and perfluorooctane sulfonic acid (PFOS)—in drinking water under the Safe Drinking Water Act. This section describes the EPA's regulatory determination findings with respect to three additional PFAS and mixtures of four PFAS.

Pursuant to sections 1412(b)(1)(A) and 1412(b)(1)(B)(ii)(II) of SDWA, the EPA is making a final determination to individually regulate as contaminants PFHxS, PFNA, and HFPO–DA and is publishing Maximum Contaminant Level Goals (MCLGs) and promulgating National Primary Drinking Water Regulations (NPDWRs) for these compounds individually. Under this authority, the EPA is also making a final determination to regulate as a contaminant a mixture of two or more of the following: perfluorononanoic acid (PFNA), hexafluoropropylene oxide dimer acid (HFPO–DA, commonly known as GenX Chemicals), perfluorohexane sulfonic acid (PFHxS), and perfluorobutane sulfonic acid (PFBS), and is publishing an MCLG and promulgating an NPDWR for mixtures of these compounds. The agency has determined that PFHxS, PFNA, and HFPO–DA may have individual adverse health effects, and any mixture of these three PFAS and PFBS may also have dose-additive adverse effects on the health of persons; that there is a substantial likelihood that PFHxS, PFNA, and HFPO–DA occur individually with a frequency and at levels of public health concern and that mixtures of these three PFAS and PFBS occur and co-occur in public water systems (PWSs) with a frequency and at levels of public health concern; and that, in the sole judgment of the Administrator, individual regulation of PFHxS, PFNA, and HFPO–DA, and regulation of mixtures of these three PFAS and PFBS, presents a meaningful opportunity for health risk reduction for persons served by PWSs. The EPA refers to ''mixtures'' in its regulatory determinations to make clear that its determinations cover all the combinations of PFHxS, PFNA, HFPO–DA, and PFBS that could co-occur in a mixture but that each regulated mixture is itself a contaminant.

While the final determination includes mixtures of PFBS in combinations with PFHxS, HFPO–DA, and PFNA, the EPA is deferring the final individual regulatory determination for PFBS to further evaluate it individually under the three SDWA regulatory determination criteria; consequently, the agency is not promulgating an individual MCLG or NPDWR for PFBS in this action. The EPA is deferring its final individual regulatory determination because after considering the public comments, the EPA has decided to further consider whether occurrence information supports a finding that there is a substantial likelihood that PFBS will individually occur in public water systems and at levels of health concern. However, as stated previously, when evaluating PFBS in mixtures combinations with PFHxS, PFNA, and/or HFPO–DA, the EPA has determined that based on the best available information it does meet all three statutory criteria for regulation when a part of these mixtures, including that it is anticipated to have dose-additive adverse health effects (see sections III.B and IV.B.1), there is a substantial likelihood of its co-occurrence in combinations with PFHxS, PFNA, and/or HFPO–DA with a frequency and at levels of public health concern (see sections III.C, VI.C, VI.D, and USEPA 2024b), and there is a meaningful opportunity for health risk reduction by regulating mixture combinations of these four PFAS (see section III.D of this preamble). Hence, although the agency is deferring the individual final regulatory determination for PFBS, it is included in the final determination to regulate mixture combinations containing two or more of PFHxS, PFNA, HFPO–DA, and PFBS.

This section describes the best available science and public health information used by the agency to support the regulatory determinations. The MCLGs and NPDWR, including the MCLs, are discussed further in sections IV and V of this preamble.

1. Proposal

The agency proposed preliminary determinations to regulate PFHxS, PFNA, HFPO–DA, and PFBS individually, and to regulate mixtures of these four PFAS contaminants, in drinking water. In the proposal, the agency concluded that PFHxS, PFNA, HFPO–DA, and PFBS, and mixtures of these PFAS, may cause adverse effects on the health of persons; there is a substantial likelihood that they will occur and co-occur in PWSs with a frequency and at levels of public health concern, particularly when considering them in a mixture; and in the sole judgment of the Administrator, regulation of PFHxS, PFNA, HFPO–DA, PFBS, and mixtures of these PFAS, presents a meaningful opportunity for health risk reductions for people served by PWSs.

Within the proposal, the agency described section 1412(b)(1)(E) which provides that the Administrator may publish a proposed drinking water regulation concurrent ''with the determination to regulate.'' This provision authorizes a more expedited process by allowing the EPA to make concurrent the regulatory determination and rulemaking processes. As a result, for the proposal, the EPA interpreted the relevant reference to ''determination to regulate'' in section 1412(b)(1)(E) as referring to the regulatory process in 1412(b)(1)(B)(ii) that begins with a preliminary determination. Under this interpretation, section 1412(b)(1)(E) authorizes the EPA to issue a preliminary determination to regulate a contaminant and a proposed NPDWR addressing that contaminant concurrently and request public comment at the same time. This allows the EPA to act expeditiously where appropriate to issue a final determination to regulate concurrently with a final NPDWR to avoid delays to address contaminants that meet the statutory criteria.

Additionally, as part of the proposal, the EPA explained why mixtures of PFAS qualify as a ''contaminant'' for purposes of section 1412. SDWA section 1401(6) defines the term ''contaminant'' to mean ''*any* physical, chemical or biological or radiological substance or matter in water.'' A mixture of two or more of the regulated PFAS qualifies as a ''contaminant'' because the mixture itself is ''*any* physical, chemical or biological or radiological substance or matter in water'' (emphasis added). Therefore, pursuant to the provisions outlined in section 1412(b)(1)(A) and 1412(b)(1)(B) of SDWA, the agency made a preliminary determination to regulate PFHxS, PFNA, HFPO–DA, PFBS, and any mixtures of these PFAS as a contaminant in drinking water. In the past and in this instance, the EPA's approach to regulating contaminant groups or mixtures under SDWA considers several factors, including health effects, similarities in physical and chemical properties, contaminant co-occurrence, ability for treatment technology co-removal, or where such a regulatory structure presents a meaningful opportunity to improve public health protection.

2. Summary of Major Public Comments and EPA Responses

The EPA requested comments on its preliminary regulatory determinations for PFHxS, PFNA, HFPO–DA, and PFBS, and mixtures of these PFAS, including the agency's evaluation of the statutory criteria and any additional data or studies the EPA should consider that inform the preliminary regulatory determinations for these contaminants and their mixtures. The EPA also requested comment on its preliminary determination that regulation of PFHxS, PFNA, HFPO–DA, PFBS, and their mixtures, in addition to regulation of PFOA and PFOS, will also provide protection from PFAS (*e.g.,* PFDA, PFDoA, PfHpA, PFHxA, PFHpS, PFPeS) that will not be regulated because the treatment technologies that would be used to ensure compliance for these PFAS are also effective in reducing concentrations of other unregulated PFAS.

Many commenters expressed support for the EPA's preliminary regulatory determinations, including that the EPA has appropriately determined that the three statutory criteria for regulation have been met for all four contaminants and their mixtures using the best available information. Many other commenters did not agree that the agency presented sufficient information to make a preliminary determination to regulate PFHxS, PFNA, HFPO–DA, PFBS, and their mixtures, with some commenters recommending that that the agency withdraw the portion of the proposed rule associated with these four PFAS because in their view there is insufficient health effects and/or occurrence data at this time to support the EPA's action. For some of the four contaminants and their mixtures, a few commenters stated that the EPA had not met the statutory criteria for regulation or that data suggests a determination not to regulate is more appropriate. The EPA disagrees with these commenters because there is information to support individual regulation of PFHxS, PFNA, and HFPO–DA, as well as mixtures of these three PFAS and PFBS, based on the three statutory criteria (these findings are discussed in this section).

As discussed earlier in this section, after consideration of all the public comments on this issue, the agency is deferring the determination to individually regulate PFBS for further evaluation under the statutory criteria. This determination is informed by public comment suggesting that the three statutory criteria for individual regulation of PFBS, particularly related to the occurrence criterion have not been met. The EPA will continue to consider other available occurrence information, including from UCMR 5, to determine whether the information supports a finding that there is a substantial likelihood that PFBS will individually occur in PWSs and at a level of public health concern. The record demonstrates that exposure to a mixture with PFBS may cause adverse health effects; that there is a substantial likelihood that PFBS co-occurs in mixtures with PFHxS, PFNA, and/or HFPO–DA in PWSs with a frequency and at levels of public health concern; and that, in the sole judgment of the Administrator, regulation of PFBS in mixtures with PFHxS, PFNA, and/or HFPO–DA presents a meaningful opportunity for health risk reduction for persons served by PWSs.

Furthermore, the EPA is making a final determination to regulate PFHxS, PFNA, and HFPO–DA individually. While the EPA recognizes there will be additional health, occurrence, or other relevant information for these PFAS and others in the future, the EPA has determined that there is sufficient information to make a positive regulatory determination and the agency concludes that these three PFAS currently meet all of the statutory criteria for individual regulatory determination. Therefore, the agency is proceeding with making final determinations to regulate these contaminants both individually and as part of mixtures with PFBS and is concurrently promulgating individual MCLs for PFHxS, PFNA, and HFPO–DA (see section V of this preamble). For detailed information on the EPA's evaluation of the three regulatory determination statutory criteria for PFHxS, PFNA, and HFPO–DA individually and mixtures of these three PFAS and PFBS, as well as more specific comments and the EPA responses related to each of the three statutory criteria, see subsections III.B, C, and D.

Several commenters requested that the EPA evaluate additional occurrence data to further inform its analysis for the regulatory determinations. In response to public comments on the proposal, the EPA evaluated updated and new occurrence data and the updates are presented within subsection III.C. and section VI of this preamble. These additional occurrence data further confirm that the SDWA criteria for regulation have been met for PFHxS, PFNA, and HFPO–DA as individual contaminants and for mixtures of PFHxS, PFNA, HFPO–DA, and/or PFBS.

A couple of commenters questioned the EPA's rationale for selecting PFHxS, PFNA, HFPO–DA, and PFBS for regulation. The agency's process is allowable under SDWA and, as described within this section of the preamble, there is available health, occurrence, and other meaningful opportunity information for three PFAS (PFHxS, PFNA, and HFPO–DA) to meet the SDWA statutory criteria for regulation individually and four PFAS (PFHxS, PFNA, HFPO–DA, and PFBS) as a mixture. The EPA disagrees with commenters who suggested that the agency should not develop national regulations that differ from state-led actions. While states may establish drinking water standards for systems in their jurisdiction prior to regulation under SDWA, once an NPDWR is in place, SDWA 1413(a)(1) requires that states or Tribes adopt standards that are no less stringent than the NPDWR to maintain primacy. Moreover, the agency further notes that all four PFAS the EPA is regulating individually or as a mixture are currently regulated by multiple states as shown in table 4–17 of USEPA, 2024e.

The EPA received several comments related to the EPA's interpretation in the proposal that the agency may, as it did here, issue a preliminary regulatory determination concurrent with a proposed NPDWR. Many stated that the EPA is authorized under SDWA to process these actions concurrently and agreed with the EPA's interpretation of the statute, noting that the EPA has followed all requirements under SDWA including notice and opportunity for public comment on both the preliminary regulatory determination and proposed NPDWR, and that simultaneous public comment periods are not precluded by SDWA. Several other commenters expressed disagreement with the EPA's interpretation. These dissenting commenters contend that the statute only allows the EPA to ‘‘publish such proposed regulation concurrent with the determination to regulate'' (*i.e.,* in their view, the final determination), not the ‘‘preliminary determination to regulate.'' Moreover, some of these commenters further indicated that they believe the EPA's final determination to regulate must precede the EPA's proposed regulation. The EPA disagrees with commenters who stated that the EPA cannot issue a preliminary determination concurrent with a proposed NPDWR. Section 1412(b)(1)(e) states that ‘‘[t]he Administrator shall propose the maximum contaminant level goals and national primary drinking water regulation for a contaminant *not later than 24 months* *after the determination to regulate under subparagraph (B),* and may publish such proposed regulation *concurrent with the determination to regulate*'' (emphasis added). The EPA maintains its interpretation that ''determination to regulate'' in the second phrase of 1412(b)(1)(E) allows for concurrent processing of a preliminary determination and proposed rule, not a final determination and proposed rule.

The first clause of the provision provides an enforceable 24-month deadline for the EPA to issue a proposed rule once it has decided to regulate. Contrary to the suggestion of some commenters, the statutory language providing that the EPA ''shall'' propose an NPDWR ''not later than 24 months *after* the determination to regulate'' states when the 24 months to issue a proposed rule begins, *i.e.,* the deadline is 24 months *after* making a final determination to issue a proposed regulation. The phrase ''after the determination to regulate'' here simply identifies when SDWA's deadline begins to run; there is no textual or other indication in the language that Congress meant it to constitute the beginning of an exclusive 24-month window in which the EPA is permitted to propose an NPDWR. Further, though the EPA's reading is clear on the face of the provision, it is also supported by language elsewhere in SDWA illustrating that when Congress intends to provide a window for action (as opposed to a deadline for action) it knows how to do so clearly. In fact, Congress did so in this very provision when it required the EPA to ''publish a maximum contaminant level goal and promulgate a national primary drinking water regulation *within* 18 months after the proposal thereof.'' See also, 42 U.S.C. 1448 (providing, among other things, that petitions for review of the EPA regulations under SDWA ''shall be filed *within* the 45-day period *beginning on* the date of the promulgation of the regulation . . .'') (emphasis added). In addition, the phrase ''not later than,'' expressly acknowledges that the EPA may issue a proposed rule concurrent with a final determination. And because this language only provides a deadline without a beginning trigger, the language in the first clause of this provision would also not preclude the EPA from issuing a proposed rule at any time prior to the expiration of the 24 months after a final regulatory determination, including issuing the proposed rule on the same day as the preliminary regulatory determination.

The second clause, which states that the Administrator ''may publish such proposed regulation concurrent with the determination to regulate'' should not be read to limit when the EPA can issue a proposed rule *prior* to a final determination. First, Congress's use of the phrase ''determination to regulate'' elsewhere in SDWA is not consistent, requiring the agency to discern its meaning based on statutory context. Second, reading ''determination to regulate'' to refer to a final determination would, without good reason, hinder Congress' goal in enacting this provision, to accelerate the EPA action under SDWA. Finally, the EPA's interpretation to allow for concurrent processes is fully consistent with, and indeed enhances, the deliberative stepwise process provided in the statute for regulating new contaminants.

Language throughout the statute demonstrates that Congress did not use the term ''determination to regulate'' consistently. In fact, ''preliminary determination'' only appears once in the entire provision, ''final determination'' is never used, and the remainder of the references simply refer to ''determination.'' Specifically, section 1412(b)(1)(B)(ii)(I) expressly requires public comment on a ''preliminary'' regulatory determination made as part of the contaminant candidate listing process. The rest of section 1412(b)(1)(B)(ii) and (iii) as well as the title of the provision only refer to a ''determination to regulate'' or ''determination.'' For example, 1412(b)(1)(B)(iii) states that ''[e]ach document setting forth the determination for a contaminant under clause (ii) shall be available for public comment at such time as the determination is published.''[1] Although this provision only refers to a ''determination for a contaminant under clause (ii),'' this language clearly refers to public comment on a preliminary determination and not a final determination to regulate. The EPA has interpretated ''determination'' in this paragraph to refer to ''preliminary determination'' because that is the best interpretation to effectuate Congressional intent to provide public comment prior to issuing a final determination. The EPA has done the same with section 1412(b)(1)(E) here, as only a reading that allows for, in appropriate cases, concurrent processing of a preliminary determination to regulate and proposed NPDWR allows for rulemaking acceleration by the EPA as Congress envisioned. To the extent there is ambiguity, the EPA's reading of section 1412(b)(1)(E) is the best one to effectuate these purposes.

The EPA could issue a proposed rule concurrent with a final determination; there is nothing in the statute or the APA that requires the EPA to wait. The SDWA gives the EPA 24 months to act after a final determination but does not require the agency to wait 24 months. The ''no later than'' language in the first clause of section 1412(b)(1)(E), expressly acknowledges that the EPA may issue a proposed rule concurrent with a final determination. Therefore, construing the second phrase of section 1412(b)(1)(E) simply to authorize the EPA to issue a proposed rule concurrent with a final determination renders that provision of the statute authorizing the EPA to publish such proposed regulation concurrent with the determination to regulate a nullity. The well-known tools of statutory construction direct the agencies and courts not to construe statutes so as to render Congress's language mere surplusage, yet that it is what commenters' interpretation would do. The EPA's construction is the one which gives meaning to that language.

Moreover, the EPA's interpretation of ''determination to regulate'' in the phrase ''may publish such proposed regulation concurrent with the determination to regulation'' in section 1412(b)(1)(E) to be a preliminary determination best effectuates Congress' goal in enacting this provision, to accelerate the EPA action under SDWA when the EPA determines such a step is necessary and the EPA has, as it does here, a sufficient record to proceed with both regulatory determination and regulation actions concurrently. In addition to authorizing concurrent processes, Congress' intent to expedite regulatory determinations when necessary is evidenced more generally by the text and structure of section 1412(b)(1)(B)(ii). The statute contemplates regulatory determinations could be made as part of the 5-year cycle for the contaminant candidate list under section 1412(b)(1)(B)(ii)(I) but may also be made at any time under section 1412(b)(1)(B)(ii)(III). The fact that Congress provided the EPA with express authority to make a regulatory determination at any time is a recognition that the EPA may need to act expeditiously to address public

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[1] Even the first clause of section 1412(b)(1)(E) setting the 24-month deadlines use ''regulatory determination'' without further clarifying whether it is preliminary or final. Again, it is clear when viewed in context that the term refers to a final determination, as triggering a deadline to propose regulations on a preliminary decision to regulate would not be reasonable, as the agency may change its mind after reviewing publicv comment, obviating the need for a proposed NPDWR.

health concerns between the statutory periodic 5-year cycle. The EPA's interpretation of the relevant language in section 1412(b)(1)(E) best effectuates all provisions of the statute because simultaneous public processes for off-cycle regulatory determinations and NPDWRs allow for administrative efficiency that may be needed to address pressing public health concerns.

Finally, the EPA's interpretation of the statute allowing for concurrent processes is fully consistent with the stepwise process for issuing an NPDWR set out by the statute. Here, the EPA provided for public comment on an extensive record for both the regulatory determinations and the proposed regulatory levels and it is not clear what further benefit would be provided by two separate public comment periods. This is especially true given the D.C. Circuit's ruling in *NRDC* v. *Regan,* 67 F.4th 397 (D.C. Cir 2023), which held that the EPA cannot withdraw a final determination to regulate a contaminant. Thus, even if the EPA were to provide two separate comment periods, the information provided on a proposed rule cannot be used to undo a final regulatory determination. Indeed, although not required by the statute, the EPA in proposing actions concurrently provides commenters with much more information to evaluate the preliminary regulatory determinations. This is because the EPA has provided not just the information to support the preliminary determinations to regulate but also the full rulemaking record and supporting risk, cost, occurrence, and benefit analysis that supports the proposed Maximum Contaminant Levels (MCLs). Further, the EPA has a much more comprehensive record for the regulatory determinations to ensure that the final determination, which cannot be withdrawn, is based on the comprehensive record provided by the rulemaking and Health Risk Reduction and Cost Analysis (HRRCA) development processes.

The EPA received comments on its statutory authority to regulate mixtures of PFHxS, PFNA, HFPO–DA, and/or PFBS, specifically the agency's interpretation under section 1401(6) that a mixture of two or more contaminants also qualifies as the definition of a contaminant under SDWA since a mixture itself meets the same definition. A few commenters disagreed and contended that a mixture does not meet the definition of being a single contaminant under SDWA. The EPA disagrees with these commenters, as the SDWA definition of a contaminant does not specify that a contaminant is only a singular chemical. The SDWA definition is very broad, specifically stating that a contaminant is ''*any* physical, chemical or biological or radiological substance or matter'' (emphasis added), with no specific description or requirement for how it is formed. Matter for example, by definition, is comprised of either pure substances or mixtures of pure substances. A pure substance is either an element or compound, which would include any PFAS chemical. The statute encompasses ''matter'' which is a broad term that includes mixtures and therefore definitionally includes PFAS mixtures, comprised of a combination of PFAS (chemical substances), as itself qualifying as a ''contaminant'' under SDWA. Moreover, other provisions of the statute, would be restricted in a manner inconsistent with Congressional intent if the EPA were to adopt the cabined approach to ''contaminant'' suggested by some commenters. For example, section 1431 of SDWA provides important authority to the EPA to address imminent and substantial endangerment to drinking water supplies posed by ''a contaminant'' that is present in or threatened those supplies. Congress clearly intended this authority to be broad and remedial, but it would be significantly hampered if the EPA would be restricted to only addressing individual chemicals and not mixtures threatening a water supply. For these reasons, the EPA's interpretation of the definition of contaminant is the only reading that is consistent with the statutory definition and use of the term in context and at to the extent the definition of contaminant is ambiguous, the EPA's interpretation represents the best interpretation of that term. Finally, even if a mixture is considered a group, as some commenters suggest, Congress clearly contemplated that the EPA could regulate contaminants as groups. *See* H.R. Rep. No 93–1185 (1974), reprinted in 1974 U.S.C.C.A.N. 6454, 6463–64) (noting the tens of thousands of chemical compounds in use commercially, with many more added each year, of which many will end up in the nation's drinking water and finding that ''[i]t is, of course, impossible for EPA to regulate each of these contaminants which may be harmful to health on a contaminant-by-contaminant basis. Therefore, the Committee anticipates that the Administrator will establish primary drinking water regulations for some groups of contaminants, such as organic and asbestos.'') Thus, the EPA has the authority to regulate a mixture as a contaminant under SDWA.

The commenters also suggested that the EPA has not followed its *Supplementary Guidance for Conducting Health Risk Assessment of Chemical Mixtures* (USEPA, 2000a), specifically that the agency did not use a ''sufficiently similar mixture'' where ''components and respective portions exist in approximately the same pattern'' and suggested that there has to be consistent co-occurrence of the mixture components. The EPA disagrees with these comments. It is not possible or necessary to use a whole-mixture approach for PFHxS, PFNA, HFPO–DA, and PFBS or a ''sufficiently similar mixture.'' Instead, the EPA is using a longstanding component-based mixture approach called the Hazard Index, which was endorsed in the context of assessing potential risk associated with PFAS mixtures by the Science Advisory Board (SAB) during its 2021 review of the EPA's *Draft Framework for Estimating Noncancer Health Risks Associated with Mixtures of Per- and Polyfluoroalkyl Substances (PFAS)* (USEPA, 2021e) (see section IV of this preamble). The goal of this component-based approach is to approximate what the whole-mixture toxicity would be if the whole mixture could be tested and relies on toxicity information for each individual component in a mixture (USEPA, 2000a). A whole-mixture approach for regulating these four PFAS in drinking water is not possible because it would entail developing a single toxicity value (*e.g.,* a reference dose (RfD)) for one specific mixture of PFHxS, PFNA, HFPO–DA, and PFBS with defined proportions of each PFAS. Toxicity studies are typically conducted with only one test substance to isolate that particular substance's effects on the test organism, and whole-mixture data are exceedingly rare. There are no known whole-mixture studies for PFHxS, PFNA, HFPO–DA, and PFBS, and even if they were available, the corresponding toxicity value (*i.e.,* a single RfD for a specific mixture of these four PFAS) would only be directly applicable to that specific mixture. Thus, a more flexible approach that takes into account the four component PFAS in different combinations and at different concentrations (*i.e.,* the Hazard Index approach) is necessary. The Hazard Index indicates risk from exposure to a mixture and is useful in this situation to ensure a health-protective MCLG in cases where the mixture is spatially and/or temporally variable. For a more detailed discussion on whole-mixture and component-based approaches for PFAS health assessment, please see the EPA's *Framework for*

Estimating Noncancer Health Risks Associated with Mixtures of Per- and Polyfluoroalkyl Substances (PFAS) (USEPA, 2024a).

Many other commenters supported the EPA's interpretation of regulating a mixture as a ''contaminant'' that consists of a combination of certain PFAS, citing the EPA's broad authority under SDWA to set regulatory standards for groups of related contaminants and the EPA precedent for doing so under other NPDWRs including disinfection byproducts (DBPs; for total trihalomethanes [TTHMs] and the sum of five haloacetic acids [HAA5] (USEPA, 1979; USEPA, 2006a)), as well as radionuclides (USEPA, 2000c) and polychlorinated biphenyls (PCBs). The EPA also noted some of these examples within the proposed rule. One commenter disagreed that these previous EPA grouping approaches are applicable to the mixture of the four PFAS, noting that TTHMs and HAA5 are byproducts of the disinfection process and are the result of naturally occurring compounds reacting with the disinfectants used in drinking water treatment; thus, their formation cannot be controlled and is dependent on the presence and amount of disinfectant. As a result of these factors, measuring them as a class is required; however, the four PFAS are not byproducts, and the presence of one PFAS does not change the presence of the other PFAS. Moreover, the commenter provided that related to radionuclides, alpha particles are identical regardless of their origination and using this example for PFAS is not supported since the four PFAS are fundamentally different. The EPA disagrees with this commenter. As noted above, the SDWA definition of contaminant is very broad (''*any* physical, chemical or biological or radiological substance or matter'' (emphasis added)) with no limitations, specific description or requirement for how it is formed. The statute therefore easily encompasses a mixture, comprised of a combination of PFAS (chemical substances), as itself qualifying as a ''contaminant'' under SDWA. Moreover, as also noted above, to the extent the mixture is considered a ''group,'' Congress clearly anticipated that the EPA would regulate contaminants by group. As a result, even if the PFAS ''group'' is different than other SDWA regulatory groupings, such a regulation is clearly authorized under the statute. Furthermore, it makes sense to treat these mixtures as a ''contaminant'' because the four PFAS share similar characteristics: it is substantially likely that they co-occur; the same treatment technologies can be used for their removal; they are measured simultaneously using the same analytical methods; they have shared adverse health effects; and they have similar physical and chemical properties resulting in their environmental persistence.

3. The EPA's Final Determination

The EPA is making determinations to regulate PFHxS, PFNA, and HFPO–DA individually and to regulate mixtures of PFHxS, PFNA, HFPO–DA, and/or PFBS. A mixture of PFHxS, PFNA, HFPO–DA, and PFBS can contain any two or more of these PFAS. The EPA refers to ''mixtures'' in its final regulatory determinations to make clear that its determinations cover all of the combinations of PFHxS, PFNA, HFPO–DA, and PFBS that could co-occur in a mixture but that any combination itself qualifies as a contaminant.

In this preamble, as discussed earlier, the EPA is deferring the final determination to regulate PFBS individually to further evaluate the three criteria specified under SDWA 1412(b)(1)(A), particularly related to its individual known or likely occurrence, but is making a final determination to regulate PFBS as part of a mixture with PFHxS, PFNA, and/or HFPO–DA.

To support the agency's regulatory determinations, the EPA carefully considered the public comments and examined health effects information from available final peer-reviewed human health assessments and studies, as well as drinking water monitoring data collected as part of the UCMR 3 and state-led monitoring efforts. The EPA finds that oral exposure to PFHxS, PFNA, and HFPO–DA individually, and combinations of these three PFAS and PFBS in mixtures, may result in a variety of adverse health effects, including similar or shared adverse effects on several biological systems including the endocrine, cardiovascular, developmental, immune, and hepatic systems (USEPA, 2024f). Based on the shared toxicity types, exposure to PFHxS, PFNA, or HFPO–DA individually, or combinations of these three PFAS and PFBS in a mixture, is anticipated to affect common target organs, tissues, or systems to produce dose-additive effects from co-exposures. Additionally, based on the agency's evaluation of the best available science, including a review of updated data from state-led drinking water monitoring efforts discussed in subsection III.C of this preamble, the EPA finds that PFHxS, PFNA, and HFPO–DA each have a substantial likelihood to occur in finished drinking water and that these three PFAS and PFBS are also likely to co-occur in mixtures and result in increased total PFAS exposure above levels of public health concern. Therefore, as discussed further in this section, the agency is determining that:

- exposure to PFHxS, PFNA, or HFPO–DA individually, and any mixture of these three PFAS and PFBS, may have adverse effects on the health of persons;
- there is a substantial likelihood that PFHxS, PFNA, and HFPO–DA will occur and there is a substantial likelihood that combinations of these three PFAS plus PFBS will co-occur in mixtures in PWSs with a frequency and at levels of public health concern; and
- in the sole judgment of the Administrator, individual regulation of PFHxS, PFNA, and HFPO–DA, and mixtures of the three PFAS plus PFBS, presents a meaningful opportunity for health risk reductions for persons served by PWSs.

The EPA is making a final individual regulatory determination for PFHxS, HFPO–DA, and PFNA and promulgating individual MCLGs and NPDWRs for PFHxS, HFPO–DA, and PFNA. These NPDWRs ensure public health protection when one of these PFAS occurs in isolation above their MCLs and also support risk communication efforts for utilities (see section V of this preamble for more information). The EPA is also making a final mixture regulatory determination and promulgating a Hazard Index MCLG and NPDWR for mixtures containing two or more of PFHxS, PFNA, HFPO–DA, and PFBS. The Hazard Index is a risk indicator and has been shown to be useful in chemical mixtures decision contexts (USEPA, 2023c).[2] Individual NPDWRs do not address dose additive risks from co-occurring PFAS. However, the Hazard Index NPDWR accounts for PFAS co-occurring in mixtures where the individual concentrations of one or more PFAS may not exceed their individual levels of public health concern, but the combined levels of these co-occurring PFAS result in an overall exceedance of the health-protective level. In this way, the Hazard Index NPDWR protects against dose-additive effects. This approach also recognizes that exposure to the PFAS included in the Hazard Index is associated with adverse health effects at differing potencies (*e.g.,* the toxicity reference value for PFHxS is lower than

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[2] Some describe the Hazard Index as an indicator of potential hazard because it does not estimate the probability of an effect; others characterize the Hazard Index as an indicator of potential risk because the measure integrates both exposure and toxicity (USEPA 2000c; USEPA, 2023c).

the one for PFBS) and that, regardless of these potency differences, all co-occurring PFAS are included in the hazard calculation (*i.e.,* the health effects and presence of lower toxicity PFAS are neither ignored nor are they over-represented). Furthermore, the approach accounts for all the different potential combinations of these PFAS that represent a potential public health concern that would not be addressed if the EPA only finalized individual NPDWRs and considered individual PFAS in isolation.

B. Statutory Criterion 1—Adverse Health Effects

The agency finds that exposure to PFHxS, PFNA, and HFPO–DA individually, and any mixture of these three PFAS and PFBS, may have an adverse effect on the health of persons. Following is a discussion of health effects information for each of these four individual PFAS and the levels at which those health effects may be adverse. The agency developed health reference levels (HRLs) for PFHxS, PFNA, HFPO–DA, and PFBS as part of its effort to identify the adverse effects each contaminant may have on the health of persons. In this instance, the EPA identified the HRL as the level below which adverse health effects over a lifetime of exposure are not expected to occur, including for sensitive populations and life stages, and allows for an adequate margin of safety. The HRLs are also used as health-based water concentrations (HBWCs) in the calculation of the Hazard Index MCLG (see section IV).

1. PFHxS

Studies have reported adverse health effects, including on the liver, thyroid, and development, after oral exposure to PFHxS (ATSDR, 2021). For a detailed discussion on adverse effects associated with oral exposure to PFHxS, please see ATSDR (2021) and USEPA (2024f).

The EPA derived the individual HRL/HBWC for PFHxS using a chronic reference value of 0.000002 (2E–06) mg/kg/day based on adverse thyroid effects (follicular epithelial hypertrophy/hyperplasia), a sensitive noncancer effect determined to be adverse and relevant to humans, observed in male rats after oral PFHxS exposure during adulthood (ATSDR, 2021; USEPA, 2024f). The EPA applied a bodyweight-adjusted drinking water intake (DWI–BW) exposure factor for adults within the general population (0.034 L/kg/day; 90th percentile direct and indirect consumption of community water, consumer-only two-day average, adults 21 years and older) and a relative source contribution (RSC) of 0.20 to calculate the HRL/HBWC (USEPA, 2024f). The HRL/HBWC for PFHxS is 10 ng/L which was used to evaluate individual occurrence of PFHxS for the final regulatory determination as discussed in section III.C of this preamble.

2. PFNA

Studies have reported adverse health effects, including on development, reproduction, immune function, and the liver, after oral exposure to PFNA (ATSDR, 2021). For a detailed discussion of adverse effects associated with oral exposure to PFNA, please see ATSDR (2021) and USEPA (2024f).

The EPA derived the HRL/HBWC for PFNA using a chronic reference value of 0.000003 (3E–06) mg/kg/day based on decreased body weight gain and impaired development (*i.e.,* delayed eye opening, delayed sexual maturation) in mice born to mothers that were orally exposed to PFNA during gestation (with presumed continued indirect exposure of offspring via lactation) (ATSDR, 2021; USEPA, 2024f). These sensitive noncancer effects were determined to be adverse and relevant to humans (ATSDR, 2021; USEPA, 2024f). The EPA applied a DWI–BW exposure factor for lactating women (0.0469 L/kg/day; 90th percentile direct and indirect consumption of community water, consumer-only two-day average) and an RSC of 0.20 to calculate the HRL/HBWC (USEPA, 2024f). The HRL/HBWC for PFNA is 10 ng/L which was used to evaluate individual occurrence of PFNA for the final regulatory determination as discussed in section III.C of this preamble.

3. HFPO–DA

Animal toxicity studies have reported adverse health effects after oral HFPO–DA exposure, including liver and kidney toxicity and immune, hematological, reproductive, and developmental effects (USEPA, 2021b). The EPA determined that there is Suggestive Evidence of Carcinogenic Potential after oral exposure to HFPO–DA in humans, but the available data are insufficient to derive a cancer risk concentration for oral exposure to HFPO–DA. For a detailed discussion of adverse effects of oral exposure to HFPO–DA, please see USEPA (2021b).

The most sensitive noncancer effects observed among the available data were the adverse effects on liver (*e.g.,* increased relative liver weight, hepatocellular hypertrophy, apoptosis, and single-cell/focal necrosis), which were observed in both male and female mice and rats across a range of exposure durations and dose levels, including the lowest tested dose levels and shortest exposure durations. The EPA derived the HRL/HBWC for HFPO–DA from a chronic oral RfD of 0.000003 (3E–06) mg/kg/day that is based on adverse liver effects, specifically a constellation of liver lesions including cytoplasmic alteration, single-cell and focal necrosis, and apoptosis, observed in parental female mice following oral exposure to HFPO–DA from pre-mating through day 20 of lactation (USEPA, 2021b). The EPA applied a DWI–BW exposure factor for lactating women (0.0469 L/kg/day; 90th percentile direct and indirect consumption of community water, consumer-only two-day average) and an RSC of 0.20 to calculate the HRL/HBWC (USEPA, 2024f). The HRL/HBWC for HFPO–DA is 10 ng/L which was used to evaluate individual occurrence of HFPO–DA for the final regulatory determination as discussed in section III.C of this preamble.

4. PFBS

Toxicity studies of oral PFBS exposure in animals have reported adverse health effects on development, as well as on the thyroid and kidneys (USEPA, 2021a). Human and animal studies evaluated other health effects following PFBS exposure including effects on the immune, reproductive, and hepatic systems and lipid and lipoprotein homeostasis, but the evidence was determined to be equivocal (USEPA, 2021a). No studies evaluating the carcinogenicity of PFBS in humans or animals were identified. The EPA concluded that there is Inadequate Information to Assess Carcinogenic Potential for PFBS and its potassium salt (K + PFBS) by any route of exposure based on the EPA's *Guidelines for Carcinogen Risk Assessment* (USEPA, 2005a). For a detailed discussion on adverse effects after oral exposure to PFBS, please see USEPA (2021a).

As noted previously, the agency is deferring the final individual regulatory determination for PFBS. For the purposes of evaluating PFBS in mixture combinations with PFHxS, PFNA, and HFPO–DA (see section III.B.5 of this preamble), the EPA derived the HRL/HBWC for PFBS from a chronic RfD of 0.0003 (3E–04) mg/kg/day that is based on adverse thyroid effects (decreased serum total thyroxine) observed in newborn mice following gestational exposure to the potassium salt of PFBS (USEPA, 2021a). The EPA applied a DWI–BW exposure factor for women of child-bearing age (0.0354 L/kg/day; 90th percentile direct and indirect consumption of community water, consumer-only two-day average) and an

RSC (relative score contribution) of 0.20 to calculate the HRL/HBWC (USEPA, 2024f). The HRL/HBWC for PFBS is 2000 ng/L.

5. Mixtures of PFHxS, PFNA, HFPO–DA, and PFBS

Exposure to per- and polyfluoroalkyl acids (PFAAs), a subclass of PFAS that includes PFHxS, PFNA, HFPO–DA, and PFBS, can disrupt signaling of multiple biological pathways, resulting in a shared set of adverse effects, including effects on thyroid hormone levels, lipid synthesis and metabolism, development, and immune and liver function (ATSDR, 2021; EFSA et al., 2018; EFSA et al., 2020; USEPA, 2021a; USEPA, 2021b; USEPA, 2024f; see further discussion in section III.B.6.e of this preamble).

Studies with PFAS and other classes of chemicals support the health-protective conclusion that chemicals that have similar observed adverse effects following individual exposure should be assumed to act in a dose-additive manner when in a mixture unless data demonstrate otherwise (USEPA, 2024a). Dose additivity means that the combined effect of the component chemicals in the mixture (in this case, PFHxS, PFNA, HFPO–DA, and/or PFBS) is equal to the sum of their individual doses or concentrations scaled for potency (USEPA, 2000a). In other words, exposure to these PFAS, at doses that individually would not likely result in adverse health effects, when combined in a mixture may result in adverse health effects. See additional discussion of PFAS dose additivity in section IV of this preamble.

The EPA used a Hazard Index (HI) HRL of 1 (unitless) to evaluate co-occurrence of combinations PFHxS, PFNA, HFPO–DA, and PFBS in mixtures for the final regulatory determination as discussed in section III.C of this preamble. For technical details on the Hazard Index approach, please see section IV of this preamble, USEPA (2024a), and USEPA (2024f).

6. Summary of Major Public Comments and EPA Responses

Commenters referred to the HRLs and HBWCs interchangeably, so comments related to those topics are addressed in this section. (Other comments related to the MCLGs are addressed in section IV of this preamble.)

Many commenters expressed support for the EPA's derivation of HRLs/HBWCs and use of best available peer-reviewed science, specifically the use of the final, most recently published Agency for Toxic Substances and Disease Registry (ATSDR) minimal risk levels for PFHxS and PFNA as chronic reference values. Other commenters criticized the EPA for using ATSDR minimal risk levels and stated that they are inappropriate for SDWA rulemaking.

The EPA finds that the ATSDR minimal risk levels for PFHxS and PFNA currently represent the best available, peer-reviewed science for these chemicals. SDWA specifies that agency actions must rely on ''the best available, peer-reviewed science and supporting studies conducted in accordance with sound and objective scientific practices.'' At this time, the 2021 ATSDR *Toxicological Profile for Perfluoroalkyls,* which covers 10 PFAS including PFHxS and PFNA, represents the best available peer-reviewed scientific information on the human health effects of PFHxS and PFNA. ATSDR minimal risk levels for PFHxS and PFNA are appropriate for use under SDWA because ATSDR uses scientifically credible approaches, its work is internally and externally peer-reviewed and undergoes public comment, and its work represents the current best available science for these two chemicals. The 2021 ATSDR *Toxicological Profile for Perfluoroalkyls* underwent intra- and interagency review and subsequent external peer review by seven experts with knowledge of toxicology, chemistry, and/or health effects.

The agency acknowledges that ATSDR minimal risk levels and EPA RfDs are not identical. The two agencies sometimes develop toxicity values for different exposure durations (*e.g.,* intermediate, chronic) and/or apply different uncertainty/modifying factors to reflect data limitations. Additionally, ATSDR minimal risk levels and EPA RfDs are developed for different purposes: ATSDR minimal risk levels are intended to serve as screening levels and are used to identify contaminants and potential health effects that may be of concern at contaminated sites, whereas EPA RfDs are used to support regulatory and nonregulatory actions, limits, and recommendations in various environmental media. However, from a practical standpoint, an oral minimal risk level and an oral RfD both represent the level of daily oral human exposure to a hazardous substance for a specified duration of exposure below which adverse health effects are not anticipated to occur. The EPA has routinely used and continues to use ATSDR minimal risk levels in human health assessments when they represent the best available science—for example, in the context of Clean Air Act section 112(f)(2) risk assessments in support of setting national emission standards for Hazardous Air Pollutants (HAPs), developing Clean Water Act ambient water quality criteria, evaluating contaminants for the CCL, and site evaluations under the Resource Conservation and Recovery Act (RCRA) and the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA).

Some commenters questioned the EPA's external peer-review process for the four underlying final toxicity assessments used to calculate the HRLs/HBWCs. Some commenters noted that the EPA does not yet have completed Integrated Risk Information System (IRIS) assessments for PFHxS and PFNA, questioning the EPA's use of non-EPA assessments (see above). The EPA notes that all four toxicity assessments containing the toxicity values (RfD or minimal risk level) used to calculate the HRLs/HBWCs (*i.e.,* the EPA human health toxicity assessments for HFPO–DA and PFBS (USEPA, 2021a; USEPA, 2021b) and the ATSDR toxicity assessments of PFNA and PFHxS (ATSDR, 2021)) underwent rigorous, external peer review (ATSDR, 2021; USEPA, 2021a; USEPA, 2021b). The EPA is not required under SDWA to exclusively use EPA assessments to support an NPDWR, and in fact, SDWA's clear direction in section 1412(b)(3)(A)(i) is to use the best *available,* peer-reviewed science when developing NPDWRs (emphasis added). Final EPA assessments for PFHxS and PFNA are under development but are not currently available; final, peer reviewed ATSDR assessments are available.

Other commenters offered critical comments on the HRLs/HBWCs for PFHxS, PFNA, HFPO–DA, and PFBS and raised technical and process concerns with the underlying human health assessments. Some commenters asserted that the human health toxicity values (EPA RfDs, ATSDR minimal risk levels) upon which the HRLs/HBWCs are based have too much uncertainty (*e.g.,* inappropriately apply a composite uncertainty factor (UF) of 3,000) and are therefore inadequate to support a SDWA regulatory determination. The EPA disagrees with these comments. The HRLs/HBWCs are data-driven values that incorporate UFs based on the EPA guidance and guidelines thus, represent the levels below which adverse health effects are not expected to occur over a lifetime. According to the EPA guidelines and longstanding practices (USEPA, 2002a; USEPA, 2022f), UFs reflect the limitations of the data across the five areas used in the current EPA human health risk assessment development: (1) human interindividual variability (UF_H); (2) extrapolation from animal to human (UF_A); (3) subchronic-to-chronic duration extrapolation (UF_S); (4) lowest-observed-adverse-effect level-to-no-observed-adverse-effect level (LOAEL-to-NOAEL) extrapolation (UF_L); and (5) database uncertainty (UF_D). In minimal risk level development, ATSDR also applies uncertainty factors as appropriate to address areas of uncertainty, with the exception of subchronic-to-chronic duration extrapolation (ATSDR, 2021). For the ATSDR minimal risk levels on which the HRLs/HBWCs for PFNA and PFHxS are based, ATSDR utilized UF_Hs, UF_As, and what ATSDR calls a modifying factor to address database deficiencies (equivalent to the EPA's UF_D) (ATSDR, 2021). The EPA carefully reviewed ATSDR's application of uncertainty and modifying factors for PFNA and PFHxS and applied additional uncertainty factors as warranted. Specifically, the EPA applied an additional UF (UF_S) for PFHxS to extrapolate from subchronic to chronic duration per agency guidelines (USEPA, 2002a) and standard practice because the critical effect was not observed during a developmental lifestage (*i.e.,* the effect was in parental male rats). A chronic toxicity value (*i.e.,* RfD, MRL) represents the daily exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime; the EPA is using a chronic toxicity value to derive the MCLG to ensure that it is set at a level at or below which no known or anticipated adverse effects on human health occur and allowing an adequate margin of safety. The EPA guidelines indicate that the composite (total) UF may be equal to or below 3,000; composite UFs greater than that represent ''excessive uncertainty'' (USEPA, 2002a; USEPA, 2022f). In the case of this final NPDWR, a composite UF of 3,000 was appropriately applied to derive toxicity values used to develop HRLs/HBWCs for two of the four PFAS (HFPO–DA and PFHxS) following peer-reviewed agency guidance and longstanding practice (see USEPA (2024f) for complete discussion of UF application for all four PFAS). The EPA has previously developed an MCLG for a chemical that had a composite UF of 3,000 applied to derive a toxicity value (*e.g.,* thallium [USEPA, 1992]). Further, a composite uncertainty factor of 3,000 has been applied in the derivation of oral RfDs for several chemicals that have been evaluated within the EPA's IRIS (Integrated Risk Information System) program (*e.g.,* fluorene, cis- and trans-1,2-dichloroethylene, 2,4-dimethylphenol; please see the EPA's IRIS program website [*https://www.epa.gov/iris*] for further information).

Some commenters opposed the EPA's application of a 20 percent RSC (relative source contribution) in the HRL/HBWC calculations and stated that it was a ''conservative default'' approach not supported by available information and that adequate exposure data exist to justify an RSC other than 20 percent (although commenters did not offer a suggested alternative RSC). The EPA disagrees with these comments. The EPA applies an RSC to account for potential aggregate risk from exposure routes and exposure pathways other than oral ingestion of drinking water to ensure that an individual's total exposure to a contaminant does not exceed the daily exposure associated with toxicity (*i.e.,* threshold level or reference dose). Application of the RSC in this context is consistent with EPA methods (USEPA, 2000d) and long-standing EPA practice for establishing drinking water MCLGs and NPDWRs (*e.g.,* see USEPA, 1989; USEPA, 2004; USEPA, 2010). The RSC represents the proportion of an individual's total exposure to a contaminant that is attributed to drinking water ingestion (directly or indirectly in beverages like coffee, tea, or soup, as well as from dietary items prepared with drinking water) relative to other exposure pathways. The remainder of the exposure equal to the RfD (or minimal risk level) is allocated to other potential exposure sources (USEPA, 2000d). The purpose of the RSC is to ensure that the level of a contaminant (*e.g.,* MCLG) in drinking water, when combined with other identified potential sources of exposure for the population of concern, will not result in total exposures that exceed the RfD (or minimal risk level) (USEPA, 2000d). This ensures that the MCLG under SDWA meets the statutory requirement that it be a level of a contaminant in drinking water at or below which no known or anticipated adverse effects on human health occur and allowing an adequate margin of safety.

To determine the RSCs for the four HRLs/HBWCs, the agency assessed the available scientific literature on potential sources of human exposure other than drinking water. The EPA conducted literature searches and reviews for each of the four HRLs/HBWCs to identify potential sources of exposure and physicochemical properties that may influence occurrence in environmental media (Deluca et al., 2022; USEPA, 2024f). Considering this exposure information, the EPA followed its longstanding, peer-reviewed Exposure Decision Tree Approach in the EPA's *Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health* (USEPA, 2000d) to determine the RSC for each PFAS. As discussed by the EPA in the Hazard Index MCLG document (USEPA, 2024f), the EPA carefully evaluated studies that included information on potential exposure to these four PFAS (PFHxS, PFNA, HFPO–DA, and PFBS) via sources other than drinking water, such as food, soil, sediment, and air. For each of the four PFAS, the findings indicated that there are significant known or potential uses/sources of exposure beyond drinking water ingestion (*e.g.,* food, indoor dust) (Box 6 in the EPA Exposure Tree; USEPA, 2000d), but that data are insufficient to allow for quantitative characterization of the different exposure sources (Box 8A in USEPA, 2000d). The EPA's Exposure Decision Tree approach states that when there are insufficient environmental and/or exposure data to permit quantitative derivation of the RSC, the recommended RSC for the general population is 20 percent (Box 8B in USEPA, 2000d). This means that 20 percent of the exposure equal to the RfD is allocated to drinking water, and the remaining 80 percent is attributed to all other potential exposure sources.

Some commenters disagreed with the bodyweight-adjusted drinking water intake (DWI–BWs) that the EPA used to calculate the HRLs/HBWCs and thought the selected DWI–BWs were too high (overly health protective). One commenter stated that the DWI–BW used in the calculation of the HRL/HBWC for HFPO–DA is inappropriate and that the EPA should have used a DWI–BW for general population adults instead of for lactating women. The EPA disagrees with this comment. To select an appropriate DWI–BW for use in derivation of the HRL/HBWC for HFPO–DA, the EPA considered the HFPO–DA exposure interval used in the oral reproductive/developmental toxicity study in mice that served as the basis for chronic RfD derivation (the critical study). In this study, parental female mice were dosed from pre-mating through lactation, corresponding to three potentially sensitive human adult life stages that may represent critical windows of HFPO–DA exposure: women of childbearing age, pregnant women, and lactating women (Table 3–63 in USEPA, 2019a). Of these three, the highest DWI–BW, for lactating women (0.0469 L/kg/day), is anticipated to be protective of the other two sensitive life stages and was used to calculate the HRL/HBWC for HFPO–DA (USEPA, 2024f).

Other commenters urged the EPA to consider infants as a sensitive life stage for PFHxS, PFNA, and PFBS and use the DWI–BW for infants to calculate the HRLs/HBWCs. The EPA disagrees with this comment. The EPA's approach to DWI–BW selection includes a step to identify the sensitive population(s) or life stage(s) (*i.e.,* those that may be more susceptible or sensitive to a chemical exposure) by considering the available data for the contaminant, including the adverse health effects observed in the toxicity study on which the RfD/minimal risk level was based (known as the critical effect within the critical or principal study). Although data gaps can complicate identification of the most sensitive population (*e.g.,* not all windows or life stages of exposure and/or health outcomes may have been assessed in available studies), the critical effect and point of departure (POD) that form the basis for the RfD (or minimal risk level) can provide some information about sensitive populations because the critical effect is typically observed at the lowest tested dose among the available data. Evaluation of the critical study, including the exposure window, may identify a sensitive population or life stage (*e.g.,* pregnant women, formula-fed infants, lactating women). In such cases, the EPA can select the corresponding DWI–BW for that sensitive population or life stage from the *Exposure Factors Handbook* (USEPA, 2019a). DWI–BWs in the *Exposure Factors Handbook* are based on information from publicly available, peer-reviewed studies, and were updated in 2019. In the absence of information indicating a sensitive population or life stage, the DWI–BW corresponding to the general population may be selected. Following this approach, the EPA selected appropriate DWI–BWs for each of the four PFAS included in the Hazard Index MCLG (see USEPA, 2024f). The EPA did consider infants as a sensitive life stage for all four PFAS; however, the agency did not select the infant DWI–BW because the exposure intervals of the critical studies supporting the chronic toxicity values did not correspond to infants. Instead, the exposure intervals were relevant to other sensitive target populations (*i.e.,* lactating women or women of childbearing age) or the general population. (See also comments related to DWI–BW selection under PFBS section III.B.6.d. of this preamble).

a. PFHxS

Some commenters noted a typographical error in the HRL/HBWC calculation for PFHxS which was reported as 9.0 ng/L in the proposal. The agency has corrected the value in this NPDWR and within the requirements under 40 CFR part 141, subpart Z. The correct HRL/HBWC for PFHxS is 10 ng/L.

Two commenters questioned the human relevance of thyroid effects (*i.e.,* changes in tissue structure (*e.g.,* enlarged cells; increased numbers of cells) in the thyroids of adult male rats) observed in the critical study used to derive the ATSDR minimal risk level and the EPA's PFHxS HRL/HBWC because, as noted in the ATSDR *Toxicological Profile for Perfluoroalkyls,* this observed effect may have been secondary to liver toxicity and, therefore, the commenters state that its significance is unclear. The EPA disagrees with this comment. SDWA requires that the EPA use ''the best available, peer reviewed science'' to inform decision making on drinking water regulations. Although there is some uncertainty regarding the selection of thyroid alterations as the critical effect (as the ATSDR toxicological profile notes), at this time, the 2021 ATSDR toxicological profile represents the best available peer reviewed scientific information regarding the human health effects of PFHxS. As the most sensitive known effect as supported by the weight of the evidence, the thyroid effect was appropriately selected by ATSDR as the critical effect. Additionally, published studies in rats have shown that PFHxS exposure results in other thyroid effects, including decreases in thyroid hormone (primarily T4) levels in serum (NTP, 2018a; Ramhøj et al., 2018). Similarly, peer-reviewed final EPA assessments of other PFAS, including PFBS (USEPA, 2021a) and perfluorobutanoic acid (PFBA) (USEPA, 2022g), have concluded that these changes in rodents are adverse and human-relevant, and appropriate for RfD derivation. Furthermore, it is appropriate to use other health protective (toxicity) values developed by other authoritative governmental agencies, including ATSDR minimal risk levels, if available, as these agencies use scientifically credible approaches and their work is peer-reviewed (the ATSDR toxicological profile underwent intra- and interagency review and external peer review by seven experts with knowledge of toxicology, chemistry, and/or health effects). The ATSDR minimal risk levels reflect the best available, peer-reviewed science.

Furthermore, the EPA's draft *IRIS Toxicological Review of Perfluorohexanesulfonic Acid (PFHxS) and Related Salts (Public Comment and External Review Draft)* (USEPA, 2023d), which is in the public domain, preliminarily provides confirmatory evidence that PFHxS significantly affects human development (emphasis added): ''Overall, the available evidence indicates that *PFHxS exposure is likely to cause thyroid and developmental immune effects in humans,* given sufficient exposure conditions. For thyroid effects, the primary supporting evidence for this hazard conclusion included evidence of decreased thyroid hormone levels, abnormal histopathology results, and changes in organ weight in experimental animals. For immune effects, the primary supporting evidence included decreased antibody responses to vaccination against tetanus or diphtheria in children.'' Although the EPA did not rely on this draft IRIS toxicological review for PFHxS in this rule, the draft is available to the public and offers confirmation that PFHxS elicits developmental effects in humans.

b. PFNA

Some commenters questioned the human relevance of developmental effects observed in PFNA animal studies (*i.e.,* decreased body weight gain, delayed eye opening, delayed sexual maturation) used to derive the ATSDR minimal risk level and the EPA's PFNA HRL/HBWC. The EPA disagrees with this comment. At this time, the 2021 ATSDR *Toxicological Profile for Perfluoroalkyls* represents the best available peer-reviewed scientific information regarding the human health effects of PFNA. In addition, according to the March 2023 *Interagency PFAS Report to Congress,* PFNA is documented to affect the developmental health domain (United States OSTP, 2023), and a recently published meta-analysis (Wright et al., 2023) specifically supports decreases in birth weight as an effect of PFNA exposure in humans. Published studies have shown that PFNA exposure results in statistically significant, dose-responsive developmental effects, including reduced fetal/pup bodyweight, reduced fetal/pup survival, changes in fetal/pup liver gene expression, increased fetal/pup liver weight, and delayed onset of puberty. Also, the EPA's 1991 *Guidelines for Developmental Toxicity Risk Assessment* (USEPA, 1991a; pp. vii-ix and pp. 1–2) cites evidence that, in the absence of clear evidence to the contrary, developmental effects observed in experimental animals are interpreted as relevant to humans.

c. HFPO–DA

A few commenters submitted critical comments related to the adverse health effects associated with exposure to HFPO–DA and how these health effects are quantified to derive the RfD in the human health toxicity assessment for HFPO–DA (USEPA, 2021b). Commenters claimed that the RfD for HFPO–DA is not scientifically sound, and cited one or more of the following reasons why: (1) the selected critical effect from the study (constellation of liver lesions) includes different liver effects that were not consistently observed across male and female mice and were not necessarily all adverse; (2) the hepatic effects in mice (the selected critical effect) are mediated by a rodent specific MOA, peroxisome proliferator-activated receptor alpha (PPARα), and therefore not relevant to humans; (3) the EPA incorporated results of a pathology working group which misapplied diagnostic criteria classifying apoptotic and necrotic lesions; and (4) the EPA misapplied uncertainty factors (UFs) (*i.e.,* the subchronic to chronic UF and database UF) according to agency guidance resulting in the maximum possible UF of 3,000 (USEPA, 2002a; USEPA, 2022f). Another commenter thought that the interspecies UF should be further increased. Also, some commenters stated that the EPA did not properly consider all available epidemiological data. These comments are addressed in this preamble.

Overall, the EPA disagrees with the commenters and maintains that the final published peer-reviewed human health toxicity assessment that derived the RfD for HFPO–DA is appropriate and sound, reflects the best available peer-reviewed science, and is consistent with agency guidance, guidelines, and best practices for human health risk assessment. Notably, the EPA sought external peer review of the toxicity assessment *twice* (USEPA, 2018b; USEPA, 2021f), released the draft toxicity assessment for public comment and provided responses to public comment (USEPA, 2021g), and engaged a seven-member pathology working group at the National Institutes of Health—an entirely separate and independent organization—to re-analyze pathology slides from two critical studies (USEPA, 2021b, appendix D), all of which supported the EPA's conclusions in the toxicity assessment, including the RfD derivation.

Regarding critical effect selection: the EPA's approach to critical effect selection for the RfD derivation considers a range of factors, including dose at which effects are observed, biological variability (which can produce differences in effects observed between sexes), and relevance of the effect(s) seen in animals to human health. The EPA maintains that selection of the constellation of liver lesions as the critical effect for HFPO–DA RfD derivation is appropriate and scientifically justified, and that the constellation of liver lesions represents an adverse effect. The EPA engaged a pathology working group within the National Toxicology Program (NTP) at the National Institutes of Health to perform an independent analysis of the liver tissue slides. The pathology working group determined that the tissue slides demonstrated a range of adverse effects and that the constellation of liver effects caused by HFPO–DA exposure, which included cytoplasmic alteration, apoptosis, single cell necrosis, and focal necrosis, constitutes an adverse liver effect in these studies (USEPA, 2021b, appendix D). The EPA evaluated the results of the pathology working group and determined that the effects were relevant to humans according to the best available science (*e.g.,* Hall et al., 2012). Additionally, the EPA convened a second independent peer-review panel of human health risk assessment experts to review the EPA's work on HFPO–DA, including critical effect selection. The panel unanimously agreed with the selection of the constellation of liver lesions as the critical effect, the adversity of this effect and its relevance to humans (USEPA, 2021f).

The commenters' assertion that the hepatic effects observed in mice are not relevant to humans because they are PPARα-mediated is unsupported. The commenter claims that one specific effect—apoptosis—can be PPARα-mediated in rodents (a pathway that some data suggest may be of limited or no relevance to humans). However, in supporting studies cited by commenters, a decrease in apoptosis is associated with a PPARα MOA, with Corton et al. (2018) stating, ''[t]he data indicate that a physiological function of PPARα activation is to increase hepatocyte growth through an increase in hepatocyte proliferation or a decrease in apoptosis or a combination of both effects'' while HFPO–DA is associated with increased apoptosis (USEPA, 2021b). Therefore, the commenter's claim that apoptosis is associated with the known PPARα MOA is unsupported. the critical study selected by the EPA, and indeed other studies as well, reported not only apoptosis but also other liver effects such as necrosis that are not associated with a PPARα MOA and therefore are relevant for human health (Hall et al., 2012). Further, according to the available criteria, effects such as cytoplasmic alteration in the presence of liver cell necrosis are considered relevant to humans (Hall et al., 2012). Additionally, commenters asserted that a 2020 study by Chappell et al. reported evidence demonstrating that the rodent liver effects are not relevant to humans, and that the EPA failed to consider this study. It is important to note that while Chappell et al. (2020) was published after the assessment's literature search cut-off date (USEPA, 2021b, appendix A; USEPA, 2022h), the EPA considered this paper initially through the Request for Correction process (USEPA, 2022h) and noted that this study specifically assessed evidence for PPARα-driven apoptosis and did not investigate other potential modes of action or types of cell death, specifically necrosis. The authors state that they could ''not eliminate the possibility that necrotic cells were also present.'' The EPA again considered Chappell et al., (2020), in addition to other studies submitted through public comment (Heintz et al., 2022; Heintz et al., 2023; Thompson et al., 2023), and determined that these studies do not fully explore a necrotic/cytotoxic MOA with Thompson et al., 2023 stating that ''there are no gene sets for assessing necrosis in transcriptomic databases.'' Critically, the commenter and these cited studies fail to recognize that increased apoptosis is a key criterion to establish a cytotoxic MOA. As outlined in the toxicity assessment (USEPA, 2021b), Felter et al., (2018) ''identified criteria for establishing a cytotoxicity MOA, which includes: . . . (2) clear evidence of cytotoxicity by histopathology, such as presence of necrosis and/or increased apoptosis.'' Overall, the EPA has determined that these studies support the mechanistic conclusions of the toxicity assessment ''that multiple MOAs could be involved in the liver effects observed after GenX chemical exposure'' including PPARα and cytotoxicity (USEPA, 2021b).

With respect to claims that the EPA misapplied diagnostic criteria classifying apoptotic and necrotic lesions: as mentioned above, the EPA engaged a pathology working group within the NTP at the National Institutes of Health to perform an independent analysis of the liver tissue slides. Seven pathologists—headed by Dr. Elmore, who was the lead author of the pathology criteria that the commenter cites (Elmore et al., 2016)—concluded that exposure to HFPO–DA caused a ''constellation of liver effects'' that included cytoplasmic alteration, apoptosis, single cell necrosis, and focal necrosis, and that this full ''constellation of lesions'' should be considered the adverse liver effect within these studies. The EPA then used the established Hall criteria (Hall et al., 2012) to determine that since liver cell death was observed, all effects, including cytoplasmic alteration, were considered adverse and relevant to humans.

The EPA disagrees with the commenters' assertion about UF application. As noted above, agency guidance (USEPA, 2002a; USEPA, 2022f) have established the appropriateness of the use of UFs to address uncertainty and account for data limitations. UFs reflect the limitations of the data across the five areas used in the current EPA human health risk assessment development (referenced above); all individual UFs that are applied are multiplied together to yield the composite or total UF. The EPA guidance dictates that although a composite UF greater than 3,000 represents ''excessive uncertainty'' (USEPA, 2002a; USEPA, 2022f), a composite UF can be equal to 3,000. For HFPO–DA, a composite UF of 3,000 was appropriately applied to account for uncertainties, including variability in the human population, database uncertainties, and possible differences in the ways in which humans and rodents respond to HFPO–DA that reaches their tissues. Furthermore, the composite UF of 3,000 and specifically the database UF and subchronic-to-chronic UF used for HFPO–DA was peer-reviewed by a panel of human health risk assessment experts, and the panel supported the application of the database UF of 10 and the subchronic-to-chronic UF of 10 (USEPA, 2021f). Additionally, a UF_A of 3 was appropriately applied, consistent with peer-reviewed EPA methodology (USEPA, 2002a), to account for uncertainty in characterizing the toxicokinetic and toxicodynamic differences between rodents and humans. As noted in the toxicity assessment for HFPO–DA (USEPA, 2021b), in the absence of chemical-specific data to quantify residual uncertainty related to toxicokinetics and toxicodynamic processes, the EPA's guidelines recommend use of a UF_A of 3.

Finally, some commenters claimed that the EPA did not consider available epidemiological evidence showing no increased risk of cancers or liver disease attributable to exposure to HFPO–DA. The EPA disagrees with this comment because the agency considered all available scientific evidence, including epidemiological studies (USEPA, 2021b). The exhibit submitted by the commenter presents an observational analysis comparing cancer and liver disease rates in North Carolina to rates in other states. It does not present the results of a new epidemiological study that included HFPO–DA exposure measures, health outcome measures, or an assessment of association between exposure and health outcome. The exhibit submitted by the commenter consists of a secondary analysis of disease rate information that was collected from various sources and does not provide new, high-quality scientific information that can be used to assess the impact of exposure to concentrations of HFPO–DA on human health.

d. PFBS

A few commenters suggested that the EPA lower the HRL/HBWC for PFBS to account for thyroid hormone disruption during early development and cited the Washington State Action Level for PFBS, which is 345 ng/L. Washington State used the same RfD (3E–04 mg/kg-d) but a higher DWI–BW to develop their Action Level as compared to the EPA's HRL/HBWC (Washington State used the 95th percentile DWI–BW of 0.174 L/kg/day for infants, whereas the EPA selected the 90th percentile DWI–BW of 0.0354 L/kg/day for women of child-bearing age). The EPA disagrees that the infant DWI–BW is more appropriate for HRL/HBWC calculation. The EPA selected the thyroid hormone outcome (decreased serum total thyroxine in newborn mice seen in a developmental toxicity study) as the critical effect in its PFBS human health toxicity assessment (USEPA, 2021a). Notably, the RfD derived from this critical effect included application of a 10X UF to account for life-stage-specific susceptibility (UF_H). To select a DWI–BW for use in deriving the HRL/HBWC for PFBS, the EPA followed its established approach of considering the PFBS exposure interval used in the developmental toxicity study in mice that was the basis for chronic RfD derivation. In this study, pregnant mice were exposed throughout gestation, which is relevant to two human adult life stages: women of child-bearing age who may be or become pregnant, and pregnant women and their developing embryos or fetuses (Table 3–63 in USEPA, 2019a). To be clear, the critical study exposed mice to PFBS only during pregnancy and not during postnatal development; newborn mice in early postnatal development, which would correspond to the human infancy life stage, were not exposed to PFBS. Of the two relevant adult stages, the EPA selected the 90th percentile DWI–BW for women of child-bearing age (0.0354 L/kg/day) to derive the HRL/HBWC for PFBS because it is the higher of the two, and therefore more health-protective. Please see additional information related to DWI–BW selection above.

Other commenters stated that the EPA's human health toxicity assessment for PFBS is overly conservative, uncertain, and that the confidence in the chronic RfD is low. The EPA disagrees with these comments. Confidence in the critical study (Feng et al., 2017) and corresponding thyroid hormone critical effect in newborn mice was rated by the EPA as 'High;' this rating was a result of systematic study evaluation and risk of bias analysis by a team of EPA experts. The Feng et al. (2017) study, the critical effect of thyroid hormone disruption in offspring, dose-response assessment, and corresponding RfD were subjected to extensive internal EPA, interagency, and public/external peer review. While confidence in the critical study was rated 'High,' the 'Low' confidence rating for the PFBS chronic RfD was in part a result of the lack of a chronic exposure duration study in any mammalian species; this lack of a chronic duration study was one of the considerations that resulted in the EPA applying a UF of 10 to account for database limitations (UF_D). Based on the EPA's human health assessment practices, the lowest confidence rating across the areas of consideration (*e.g.,* existent hazard/dose-response database) is assigned to the corresponding derived reference value (*e.g.,* RfD). Thus, the EPA has high confidence in the critical study (Feng et al., 2017) and critical effect/thyroid endpoint, but the database is relatively limited. Although the PFBS RfD was based on best available peer-reviewed science, there is uncertainty as to the hazard profile associated with PFBS after prolonged (*e.g.,* lifetime) oral exposure. In the toxicity assessment for PFBS (USEPA, 2021a), the EPA noted data gaps in specific health effects domains, as is standard practice. Toxicity assessments for most chemicals identify data gaps; the issue of uncertainty due to toxicological study data gaps is not unique to PFBS. Data gaps are considered when selecting the UF_D because they indicate the potential for exposure to lead to adverse health effects at doses lower than the POD derived from the assessment's critical study. There is a potential that effects with greater dose-response sensitivity (*i.e.,* occurring at lower daily oral exposures) might be discovered from a chronic duration exposure study. Due to this uncertainty, the EPA applied a UF_D of 10.

One commenter questioned the EPA's approach to estimating the human equivalent dose (HED) from the animal data using toxicokinetic (TK) data rather than using default body-weight scaling and suggested that the default allometric approach is more appropriate for estimating an HED. The EPA disagrees with this comment. In human health risk assessment practice, the EPA considers a hierarchical approach to cross-species dosimetric scaling consistent with technical guidance to calculate HEDs (USEPA, 2011; see pp. X–XI of the Executive Summary in '*Recommended Use of Body Weight$^{3/4}$ as the Default Method in Derivation of the Oral Reference Dose'*). The preferred approach is physiologically based toxicokinetic (PBTK) modeling; however, there are rarely sufficient chemical-specific data to properly parameterize such a model. In the absence of a PBTK model, the EPA considers an intermediate approach in which chemical-specific data across species, such as clearance or plasma half-life, are used to calculate a dosimetric adjustment factor (DAF) (USEPA, 2011). If chemical-specific TK data are not available, only then is a default approach used wherein allometric scaling, based on body weight raised to the ¾ power, is used to calculate a DAF. The human health toxicity assessment for PFBS invoked the intermediate approach, consistent with guidance, as TK data were available for humans and rodents.

e. Mixtures of PFHxS, PFNA, HFPO–DA, and PFBS

Comments on the EPA's preliminary regulatory determination on the mixtures of PFHxS, PFNA, HFPO–DA, and/or PFBS were varied. Many commenters supported the EPA's proposal to regulate a mixture of these PFAS and agreed with the EPA's scientific conclusions about PFAS dose additivity. Many commenters urged the EPA to consider making a determination to regulate for additional PFAS (in a mixture) or all PFAS as a class. As described throughout section III of this preamble, the agency is required to demonstrate a contaminant meets the SDWA statutory criteria to make a regulatory determination. In this preamble, in addition to PFOA and PFOS which the EPA has already made a final determination to regulate, the agency is making final determinations for all PFAS with sufficiently available information to meet these statutory criteria either individually and/or as part of mixture combinations. As information becomes available, the agency will continue to evaluate other PFAS for potential future preliminary regulatory determinations.

Many commenters opposed the EPA's conclusion about PFAS dose additivity and use of the Hazard Index approach to regulate co-occurring PFAS. A few commenters agreed with the EPA's decision to regulate mixtures of certain PFAS and the EPA's conclusion about dose additivity but questioned the EPA's use of the general Hazard Index, and instead, suggested alternative approaches. Please see section IV of this preamble for a summary of comments and the EPA responses on the Hazard Index MCLG and related topics.

There is substantial evidence that PFHxS, PFNA, HFPO–DA, and PFBS act in a dose additive manner, that these four PFAS elicit similar health effects, and that exposure to mixtures of these PFAS may have adverse health effects. Following is a discussion of dose additivity and similarity of adverse effects of PFHxS, PFNA, HFPO–DA, and PFBS.

As noted in this section, the available data indicate that PFHxS, PFNA, HFPO–DA, and PFBS, while not necessarily toxicologically identical, elicit many of the same or similar adverse health effects across different levels of biological organization, tissues/organs, lifestages, and species (ATSDR, 2021; EFSA et al., 2018; EFSA et al., 2020; USEPA, 2021d; USEPA, 2021f; USEPA, 2024f). Each of these PFAS disrupts signaling of multiple biological pathways, resulting in a shared set of adverse effects including effects on thyroid hormone levels, lipid synthesis and metabolism, development, and immune and liver function (ATSDR, 2021; EFSA et al., 2018; EFSA et al., 2020; USEPA, 2021d; USEPA, 2021f; USEPA, 2024f). Please also see USEPA (2024a) for an overview of recent studies that provide supportive evidence of similar effects of PFAS.

Available health effects studies indicate that PFAS mixtures act in a dose-additive manner when the individual components share some health endpoints/outcomes. Individual PFAS, each at doses that are not anticipated to result in adverse health effects, when combined in a mixture may result in adverse health effects. Dose additivity means that when two or more of the component chemicals (in this case, PFHxS, PFNA, HFPO–DA, and/or PFBS) exist in one mixture, the risk of adverse health effects following exposure to the mixture is equal to the sum of the individual doses or concentrations scaled for potency (USEPA, 2000a). Thus, exposure to these PFAS, at doses that individually would not likely result in adverse health effects, when combined in a mixture may pose health risks.

Many commenters supported the EPA's scientific conclusions about PFAS dose additivity and agreed that considering dose-additive effects is a health-protective approach. Many other commenters disagreed with the EPA's scientific conclusions regarding PFAS dose additivity and a few commenters questioned the agency's external peer-review process and whether the agency sufficiently responded to SAB (Science Advisory Board) comments. For example, these commenters stated that the evidence base of PFAS mixture studies is too limited to support dose additivity for these four PFAS and recommended that the EPA re-evaluate its conclusion about dose additivity as new data become available. A few commenters stated that the EPA failed to adequately follow the SAB recommendation that ''discussion of studies of toxicological interactions in PFAS mixtures in the EPA mixtures document be expanded to also include studies that do not indicate dose additivity and/or a common MOA [mode of action] for PFAS.'' The EPA's responses to these comments are summarized in this section.

The EPA continues to support its conclusion that PFAS that elicit similar adverse health effects following individual exposure should be assumed to act in a dose-additive manner when in a mixture unless data demonstrate otherwise. Numerous published studies across multiple chemical classes, biological effects, and study designs support a dose-additive mixture assessment approach for PFAS because they demonstrate that experimentally observed responses to exposure to PFAS and other chemical mixtures are consistent with modeled predictions of dose additivity (see the EPA's *Framework for Estimating Noncancer Health Risks Associated with Mixtures of Per- and Polyfluoroalkyl Substances (PFAS)* (USEPA, 2024a)). Since the EPA's draft PFAS Mixtures Framework underwent SAB review in 2021, new studies from the EPA and others have published robust evidence of combined toxicity of PFAS in mixtures, corroborating and confirming earlier findings (*e.g.,* Conley et al., 2022a; Conley et al., 2022b; USEPA, 2023c; see USEPA, 2024a for additional examples). Additionally, the National Academies of

Sciences, Engineering, and Medicine (NASEM, 2022) recently recommended that clinicians apply an additive approach for evaluating patient levels of PFAS currently measured in the National Health and Nutrition Examination Survey (NHANES) in order to protect human health from additive effects from PFAS co-exposure.

The EPA directly asked the SAB for feedback on PFAS dose additivity in the charge for the 2021 review of the EPA's draft PFAS Mixtures Framework. Specifically, the EPA asked the SAB to, ''[p]lease comment on the appropriateness of this approach for a component-based mixture evaluation of PFAS under an assumption of dose additivity'' (USEPA, 2022i). The SAB strongly supported the scientific soundness of this approach when evaluating PFAS and concurred that it was a health protective conclusion. For example, the SAB said:

> . . . The information included in the draft framework supports the conclusion that toxicological interactions of chemical mixtures are frequently additive or close to additive. It also supports the conclusion that dose additivity is a public health protective assumption that typically does not underestimate the toxicity of a mixture . . . (USEPA, 2022i)
>
> The SAB Panel agrees with use of the default assumption of dose additivity when evaluating PFAS mixtures that have similar effects and concludes that this assumption is health protective. (USEPA, 2022i)

Regarding the commenters' assertion that the agency did not adequately follow the SAB recommendation to expand its discussion of PFAS mixtures study results that did not show evidence of dose additivity and/or a common MOA, the EPA disagrees. The EPA reviewed all studies provided by the SAB and in response, included a discussion of relevant additional studies in its public review draft PFAS Mixtures Framework (see section 3 in USEPA, 2023w). Since then, the EPA has included additional published studies and those findings further confirm dose additive health concerns associated with PFAS mixtures (see section 3 in USEPA, 2024a). Data from *in vivo* studies that rigorously tested accuracy of Dose Additivity (DA), Integrated Addition (IA), and Response Additivity (RA) model predictions of mixtures with components that disrupted common pathways demonstrated that DA models provided predictions that were better than or equal to IA and RA predictions of the observed mixture effects (section 3.2 in USEPA, 2024a). The National Academy of Sciences (NAS) conclusions on phthalates (and related chemicals) (NRC, 2008) and systematic reviews of the published literature (Boobis et al., 2011 and Martin et al., 2021; see also section 3.2 in USEPA, 2024a) support DA as the default model for estimating mixture effects in some circumstances, even when the mixtures included chemicals with diverse MOAs (but common target organs/effects) (Boobis et al., 2011; Martin et al., 2021; USEPA, 2024a). Recent efforts to investigate *in vitro* and *in vivo* PFAS mixture effects have provided robust evidence that PFAS behave in a dose-additive manner (see section 3 in USEPA, 2024a).

As supported by the best available science, the SAB, the agency's chemical mixtures guidance (USEPA, 1991b; USEPA, 2000a), and the EPA Risk Assessment Forum's *Advances in Dose Addition for Chemical Mixtures: A White Paper* (USEPA, 2023c), the EPA proposed a Hazard Index MCLG for a mixture of up to four PFAS (PFHxS, PFNA, HFPO–DA, and PFBS) based on dose additivity because published studies show that exposure to each of these individual four PFAS elicits some of the same or similar adverse health effects/outcomes. As noted above, many commenters, as well as the SAB (USEPA, 2022i), supported this conclusion of dose additivity based on similarity of adverse effects.

While the SAB also noted that there remain some questions about PFAS interaction in mixtures (USEPA, 2022i), the available data justify an approach that accounts for PFAS dose additivity. Studies that have assessed PFAS mixture-based effects do not offer evidence for synergistic/antagonistic effects (USEPA, 2024a). For example, Martin et al. (2021), following a review of more than 1,200 mixture studies (selected from > 10,000 reports), concluded that there was little evidence for synergy or antagonism among chemicals in mixtures and that dose additivity should be considered as the default. Experimental data demonstrate that PFAS disrupt signaling in multiple biological pathways resulting in common adverse effects on several of the same biological systems and functions including thyroid hormone signaling, lipid synthesis and metabolism, developmental toxicity, and immune and liver function (USEPA 2024a). Additionally, several EPA Office of Research and Development (ORD) studies provide robust evidence that PFAS behave in a dose-additive manner (Conley et al., 2022a; Conley et al., 2022b; Conley et al., 2023; Gray et al., 2023).

Several commenters opposed the conclusion of dose additivity based on similarity of adverse effects and stated that the EPA failed to establish that the four PFAS included in the Hazard Index (PFHxS, PFNA, HFPO–DA, and PFBS) elicit similar adverse health effects. The EPA disagrees with these comments because the available epidemiology and animal toxicology studies demonstrate that these four PFAS (PFHxS, PFNA, HFPO–DA, and PFBS) have multiple health endpoints and outcomes in common (USEPA, 2024f). Further, these four PFAS are well-studied PFAS for which the EPA or ATSDR have developed human health assessments and toxicity values (*i.e.,* RfDs, minimal risk levels). As shown in Table 1, available animal toxicological data and/or epidemiological studies demonstrate that PFHxS, PFNA, HFPO–DA, and PFBS are documented to affect at least five (5) of the same health outcomes for this evaluation: lipids, developmental, immune, endocrine, and hematologic (USEPA, 2024g). Similarly, according to the 2023 Interagency PFAS Report to Congress (United States OSTP, 2023), available animal toxicological data show that PFHxS, PFNA, HFPO–DA, and PFBS are documented to significantly affect at least eight (8) of the same major health effect domains: body weight, respiratory, hepatic, renal, endocrine, immunological, reproductive, and developmental. In short, multiple evaluation efforts have clearly demonstrated that each of the PFAS regulated by this NPDWR impact numerous of the same or similar health outcomes or domains.

Table 1: Affected health outcomes in animal toxicity and/or epidemiological studies for the four PFAS included in the Hazard Index MCLG (adapted from Table 6-7 in USEPA, 2024g)

Health Outcome	PFNA	PFHxS	PFBS	HFPO-DA
Lipids	X	X	X	X
Developmental	X	X	X	X
Hepatic	X	X	-	X
Immune	X	X	X	X
Endocrine	X	X	X	X
Renal	-	-	X	X
Hematologic	X	X	X	X

Notes: (X) Health outcome examined, evidence of association; (-) health outcome examined, no evidence of association.

In summary, there is substantial evidence that mixtures of PFHxS, PFNA, HFPO–DA, and PFBS act in a dose-additive manner and elicit multiple similar toxicological effects. Studies by the EPA and others provide evidence that corroborates the dose-additive toxicity of PFAS mixtures, and data on different chemical classes and research also provide support for dose additivity. Additionally, numerous *in vivo* and *in vitro* studies demonstrate that these four PFAS share many common health effects across diverse health outcome categories (*e.g.,* developmental, immunological, and endocrine effects), and that they induce some of the same effects at the molecular level along biological pathways (USEPA, 2024f).

C. Statutory Criterion 2—Occurrence

The EPA has determined that there is a substantial likelihood that PFHxS, PFNA, and HFPO–DA will individually occur and combinations of these three PFAS and PFBS will co-occur in mixtures in PWSs with a frequency and at levels of public health concern based on the EPA's evaluation of the best available occurrence information. In this preamble, while the EPA is making a final determination to regulate PFBS in mixtures with PFHxS, PFNA, and/or HFPO–DA, the agency is deferring the final individual regulatory determination for PFBS so that the agency can continue to evaluate this contaminant relative to the SDWA criteria for regulation, particularly related to its individual known or likely occurrence. For the other three PFAS, the EPA is making a final determination to regulate them individually in this preamble (*i.e.,* PFHxS, PFNA, and HFPO–DA). The EPA recognizes there will be additional occurrence or other relevant information for these and other PFAS in the future. The EPA has, however, determined that there is more than sufficient occurrence information to satisfy the statutory criterion to regulate PFNA, PFHxS, and HFPO–DA.

The EPA's evaluation of the second statutory criterion for regulation of PFHxS, PFNA, and HFPO–DA individually and regulation of combinations of these PFAS and PFBS in mixtures follows a similar process to previous rounds of regulatory determinations including the written Protocol developed under Regulatory Determination 3 (USEPA, 2014a) and also described in detail in the Preliminary Regulatory Determination 4 (USEPA, 2020a). Using the Protocol, and as conducted for the regulatory determinations in this action, the agency compares available occurrence data relative to the contaminant HRL, a health-based concentration against which the agency evaluates occurrence data when making regulatory determinations, as a preliminary factor in informing the level of public health concern. For both this regulatory determination and previous regulatory determinations, this is the first screening factor in informing if there is a substantial likelihood the contaminant will occur at a frequency and level of public health concern. Consistent with the Protocol and similar to all past regulatory determinations, these regulatory determinations are also based on other factors, not just the direct comparison to the HRL. As described clearly in the proposal, the EPA has not been able to determine a simple threshold of public health concern for all contaminants the agency considers for regulation under SDWA; rather, it is a contaminant-specific decision which ''involves consideration of a number of factors, some of which include the level at which the contaminant is found in drinking water, the frequency at which the contaminant is found and at which it co-occurs with other contaminants, whether there is an sustained upward trend that these contaminant will occur at a frequency and at levels of public health concern, the geographic distribution (national, regional, or local occurrence), the impacted population, health effect(s), the potency of the contaminant, other possible sources of exposure, and potential impacts on sensitive populations or lifestages.'' (USEPA, 2023f). It also includes consideration of production and use trends and environmental fate and transport parameters which may indicate that the contaminant would persist and/or be mobile in water. Appropriately, the EPA has considered these relevant factors in its evaluation that there is a substantial likelihood that PFHxS, PFNA, and HFPO–DA will individually occur and combinations of these three PFAS and PFBS will co-occur in mixtures in PWSs with a frequency and at levels of public health concern.

The EPA's evaluation of the second statutory criterion is based on the best available health information, which includes UCMR 3 data and more recent PFAS drinking water data collected by several states. Based on suggestions in public comments to update state occurrence data, the EPA supplemented the data used to inform the rule proposal with new data from states included in the original proposal and additional states that have made monitoring data publicly available since the rule proposal (USEPA, 2024b). Consistent with section 1412(b)(1)(B)(II), this information combined represents best available occurrence data. It includes results from tens of thousands of samples and the assembled data represent one of the most robust occurrence datasets ever used to inform development of a drinking water regulation of a previously unregulated contaminant. The state data were primarily gathered after the UCMR 3 using improved analytical methods that could measure more PFAS at lower concentrations. These additional data demonstrate greater occurrence and co-occurrence of the PFAS monitored under UCMR 3 (PFHxS, PFNA, and PFBS) at significantly greater frequencies than UCMR 3 and the data initially included in the analysis. Furthermore, the state data show the co-occurrence of PFAS at levels of public health concern, as well as the demonstrated occurrence and co-occurrence of HFPO–DA which was not included within UCMR 3. As discussed subsequently, these data demonstrate that there is a substantial likelihood PFHxS, PFNA, and HFPO–DA will occur and combinations of PFHxS, PFNA, HFPO–DA, and PFBS will co-occur in mixtures with a frequency and at levels of public health concern. When determining that there is a substantial likelihood PFHxS, PFNA, and HFPO–DA will occur and PFHxS, PFNA, HFPO–DA, and/or PFBS will co-occur at levels of public health concern, the EPA considered both the occurrence concentration levels for PFHxS, PFNA, and HFPO–DA individually, as well as their collective co-occurrence and corresponding dose additive health concerns from co-exposures with PFBS for purposes of considering a regulatory determination for mixtures of these four PFAS. The EPA also considered other factors in evaluating the second criterion and informing level of public health concern for PFHxS, PFNA, and HFPO–DA individually and combinations of these three PFAS and PFBS in mixtures, including the frequency at which the contaminant is found, the geographic representation of the contaminant's occurrence, and the environmental fate and transport characteristics of the contaminant. As the EPA noted previously, while the agency is not making an individual regulatory determination for PFBS at this time, PFBS is an important component in mixtures with PFHxS, PFNA, and HFPO–DA and the EPA presents occurrence information for PFBS as part of section III.C.5 and its co-occurrence analyses in sections VI.C and D of this preamble.

The EPA focused the evaluation of the state data on the non-targeted or non-site specific (*i.e.,* monitoring not conducted specifically in areas of known or potential contamination) monitoring efforts from 19 states. Non-targeted or non-site-specific monitoring is likely to be more representative of general occurrence because its framework and monitoring results will be less likely to potentially over-represent concentrations at locations of known or suspected contamination. Sixteen (16) of 19 states reported detections of at least three of PFHxS, PFNA, HFPO–DA, or PFBS.

The EPA considered the targeted state monitoring data separately since a higher rate of detections may occur as a result of specifically looking in areas of suspected or known contamination. For the targeted state data nearly all these states also reported detections at systems serving millions of additional people, as well as at levels of public health concern, both individually for PFHxS, PFNA, and HFPO–DA, and as mixtures of these three PFAS and PFBS. State data detection frequency and concentration results vary for PFHxS, PFNA, HFPO–DA, and PFBS, both between these four different PFAS and across different states, with some states showing much higher reported detections and concentrations of these PFAS than others. The overall results demonstrate the substantial likelihood that individually PFHxS, PFNA, and HFPO–DA and mixtures of these three PFAS with PFBS will occur and co-occur at frequencies and levels of public health concern. Tables 2 and 3 show the percent of samples with state reported detections of PFHxS, PFNA, HFPO–DA, and PFBS, and the percentage of monitored systems with detections of PFHxS, PFNA, HFPO–DA, and PFBS, respectively, across the non-targeted state finished water monitoring data. The EPA notes that Alabama is not included in Tables 2 and 3 as only detections were reported and there was no information on the total number of samples collected to determine percent detection.

Table 2. Non-Targeted State PFAS Finished Water Data – Summary of Samples with State Reported Detections[1] of PFHxS, PFNA, HFPO-DA, and PFBS

State	**PFHxS**	**PFNA**	**PFBS**	**HFPO-DA**
Colorado	10.8%	0.9%	11.0%	0.2%
Illinois	13.4%	0.6%	17.6%	0.0%
Indiana	1.5%	0.2%	5.6%	0.0%
Kentucky	8.6%	2.5%	12.3%	13.6%
Maine	3.0%	3.5%	10.1%	N/A[2]
Maryland	18.2%	2.3%	19.3%	0.0%
Massachusetts	23.6%	2.9%	39.8%	0.1%
Michigan	4.3%	0.6%	7.5%	0.1%
Missouri	3.3%	0.0%	6.1%	0.0%
New Hampshire	16.8%	3.3%	32.1%	3.8%
New Jersey	26.2%	7.7%	28.1%	N/A[2]
New York	21.6%	8.6%	28.8%	0.7%
North Dakota	5.3%	0.0%	8.8%	0.0%
Ohio	6.6%	0.3%	5.0%	0.1%
South Carolina	8.1%	0.1%	13.7%	1.3%
Tennessee	0.0%	0.0%	0.0%	N/A[2]
Vermont	4.2%	2.5%	7.1%	0.2%
Wisconsin	27.2%	2.2%	28.0%	0.0%

Notes:

[1] Detections determined by individual state reported limits which are not defined consistently across all states.

[2] N/A indicates the analyte was not sampled as part of the state monitoring.

Table 3: Non-Targeted State PFAS Finished Water Data – Summary of Monitored Systems with State Reported[1] Detections of PFHxS, PFNA, HFPO-DA, and PFBS

State	PFHxS	PFNA	PFBS	HFPO-DA
Colorado	13.4%	1.0%	13.4%	0.3%
Illinois	4.6%	0.5%	8.0%	0.0%
Indiana	1.3%	0.3%	6.5%	0.0%
Kentucky	9.5%	2.7%	13.5%	12.2%
Maine	2.8%	3.9%	10.3%	N/A[2]
Maryland	12.7%	3.2%	12.7%	0.0%
Massachusetts	18.1%	4.4%	27.8%	0.3%
Michigan	4.1%	0.6%	7.9%	0.3%
Missouri	2.7%	0.0%	6.2%	0.0%
New Hampshire	22.5%	5.5%	38.1%	5.1%
New Jersey	32.9%	16.5%	35.2%	N/A[2]
New York	25.0%	9.7%	36.7%	1.1%
North Dakota	5.4%	0.0%	9.0%	0.0%
Ohio	2.2%	0.3%	2.4%	0.1%
South Carolina	13.7%	0.3%	22.1%	2.0%
Tennessee	0.0%	0.0%	0.0%	N/A[2]
Vermont	2.7%	0.9%	6.0%	0.5%
Wisconsin	31.8%	3.9%	33.9%	0.0%

Notes:

[1] Detections determined by individual state reported limits which are not defined consistently across all states.

[2] N/A indicates the analyte was not sampled as part of the state monitoring.

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As shown in Tables 2 and 3, all states except three report sample and system detections for at least three of the four PFAS. For those states that reported detections, the percentage of samples and systems where these PFAS were found ranged from 1 to 39.8 percent and 0.1 to 38.1 percent, respectively. While these percentages show occurrence variability across states, several of these states demonstrate that a significant number of samples (*e.g.,* detections of PFHxS in 26.2 percent of New Jersey samples) and systems (*e.g.,* detections of HFPO–DA in 12.2 percent of monitored systems in Kentucky) contain some or all four PFAS. This occurrence information, as well as the specific discussion related to individual occurrence for PFHxS, PFNA, and HFPO–DA and co-occurrence of these three PFAS and PFBS, supports the agency's determination that there is a substantial likelihood that PFHxS, PFNA, HFPO–DA occur and PFHxS, PFNA, HFPO–DA, and PFBS co-occur in combinations of mixtures with a frequency of public health concern. Additionally, the agency emphasizes that occurrence and co-occurrence of these PFAS is not only at a regional or local level, rather it covers many states throughout the country; therefore, a national level regulation is necessary to ensure all Americans served by PWSs are equally protected.

1. PFHxS

The occurrence data presented above, throughout section VI of this preamble and discussed in the USEPA (2024b) support the agency's final determination that there is a substantial likelihood PFHxS occurs with a frequency and at levels of public health concern in drinking water systems across the United States. PFHxS was found under UCMR 3 in approximately 1.1 percent of systems, serving 5.7 million people across 25 states, Tribes, and U.S. territories. However, under UCMR 3, the minimum reporting level for PFHxS was 30 ng/L. As this reporting level is three times greater than the health-based HRL for PFHxS (10 ng/L), it is extremely likely there is significantly greater occurrence and associated population exposed in the range between the HRL of 10 ng/L and the UCMR 3 minimum reporting level of 30 ng/L (as demonstrated by both the more recent state data and the EPA's occurrence model discussed in this section and in section VI of this preamble showing many results in this concentration range). Through analysis of available state data, which consisted of approximately 48,000 samples within 12,600 systems, 18 out of the 19 states that conducted non-targeted monitoring had reported detections of PFHxS in 1.3 to 32.9 percent of their systems (Tables 2 and 3). These same systems reported concentrations ranging from 0.2 to 856 ng/L with median sample concentrations ranging from 1.17 to 12.1 ng/L, demonstrating concentrations above the HRL of 10 ng/L.

Targeted state monitoring data of PFHxS show similar results. For example, in its targeted monitoring efforts, California reported 38.5 percent of monitored systems found PFHxS, where concentrations ranged from 1.1 to 160 ng/L, also demonstrating concentrations above the HRL. In total, considering both the non-targeted and targeted state data, PFHxS was found above the HRL in at least 184 PWSs in 21 states serving a population of approximately 4.3 million people.

The EPA also evaluated PFHxS in a national occurrence model that has been developed and utilized to estimate national-scale PFAS occurrence for four PFAS that were included in UCMR 3 (Cadwallader et al., 2022). The model has been peer reviewed and is described extensively in Cadwallader et al. (2022). The model and results are described in section VI.E of this preamble; briefly, both the UCMR 3 and some state data were incorporated into a Bayesian hierarchical model which supported exposure estimates for select PFAS at lower levels than were measured under UCMR 3. Hundreds of systems serving millions of people were estimated to have mean concentrations exceeding the PFHxS HRL (10 ng/L). Therefore, the UCMR 3 results, the national occurrence model results, and the substantial state data demonstrate the substantial likelihood PFHxS occurs at a frequency and level of public health concern. Finally, UCMR 5 data are being reported to the EPA while this final rule is being prepared. See section VI of this preamble for more information on the preliminary results. While these UCMR 5 PFHxS data are too preliminary to provide the basis for the regulatory determination, these preliminary UCMR 5 results appear to confirm state data and model results.

Further supporting this final determination, PFHxS is very stable and persistent in the environment. While PFHxS was phased out in the U.S. in the early 2000's there are still detections as previously demonstrated. In addition, legacy stocks may also still be used, production continues in other countries, and products containing PFHxS may be imported into the U.S. (USEPA, 2000b). Since PFHxS is environmentally persistent and products containing PFHxS are still in use and may be imported into the United States, the EPA anticipates environmental contamination to sources of drinking water will continue. To illustrate this point further, PFOA and PFOS, two of the most extensively sampled PFAS, are also very environmentally persistent and have similarly been phased out in the U.S. for many years, though these two contaminants continue to often be found at levels of public health concern as discussed in section VI of this preamble. Currently, this also appears to be a similar trend for PFHxS occurrence, where the drinking water sample data demonstrates it continues to occur at levels of public health concern. Therefore, in consideration of factors relating to the environmental persistence of PFHxS, its presence in consumer products and possible continued use, and the observed occurrence trend of PFOA and PFOS, the EPA finds that there is a substantial likelihood PFHxS occurs or will occur at a frequency and level of public health concern.

2. PFNA

The occurrence data presented above, throughout section VI of this preamble, and discussed in USEPA (2024b) support the agency's final determination that there is a substantial likelihood PFNA occurs with a frequency and at levels of public health concern in drinking water systems across the U.S.

PFNA was found under UCMR 3 in approximately 0.28 percent of systems, serving 526,000 people in 7 states, Tribes, and U.S. territories, using a minimum reporting level of 20 ng/L. As this reporting level is two times greater than the health-based HRL of 10 ng/L, the EPA expects there is even greater occurrence and exposed population in the range between 10 and 20 ng/L. Additionally, through analysis of the extensive amount of available state data, which consisted of approximately 57,000 samples within approximately 12,400 systems, 16 of 19 non-targeted monitoring states reported detections of PFNA within 0.3 to 16.5 percent of their systems (Tables 2 and 3). These same states reported sample results ranging from 0.23 to 330 ng/L, demonstrating levels above the HRL of 10 ng/L, with median sample results ranging from 0.35 to 7.5 ng/L.

Targeted state monitoring data of PFNA are also consistent with non-targeted state data; for example, Pennsylvania reported 5.8 percent of monitored systems found PFNA, where concentrations ranged from 1.8 to 18.1 ng/L, also showing concentrations above the HRL. When considering all available state data, there are at least 480 systems in 19 states serving more than 8.4 million people that reported any concentration of PFNA, and at least 52 systems in 12 states within different geographic regions serving a population of 177,000 people with reported concentrations above the HRL of 10 ng/L. Furthermore, when evaluating only a subset of the available state data representing non-targeted monitoring, PFNA was reported in approximately 3.6 percent of monitored systems; if these results were extrapolated to the nation and those system subject to the final rule requirements, the agency estimates that PFNA would be detectable in over 2,300 PWSs serving 24.9 million people. If those results were further compared to the HRL for PFNA (10 ng/L), PFNA would be detected above the HRL in 228 systems with 830,000 people exposed. Thus, in addition to the UCMR 3 results, these extensive state data also reflect there is a substantial likelihood PFNA occurs at a frequency and level of public health concern because it is observed or likely to be observed within numerous water systems above levels of public health concern across a range of geographic locations. Finally, UCMR 5 data are being reported to the EPA while this final rule is being prepared. See section VI of this preamble for more information on the preliminary results. While these PFNA UCMR 5 data are too preliminary to provide the basis for the regulatory determination, these preliminary UCMR 5 results appear to confirm state data discussed above.

Further supporting this final determination, PFNA is very stable and persistent in the environment. While it has generally been phased out in the U.S. there are still detections as demonstrated previously. Additionally, legacy stocks may still be used and products containing PFNA may still be produced internationally and imported to the U.S. (ATSDR, 2021). Since PFNA is environmentally persistent and products containing PFNA are still in use and may be imported into the U.S., there is a substantial likelihood that environmental contamination of sources of drinking water will continue. To illustrate this point further, PFOA and PFOS, two of the most extensively sampled PFAS, are also very environmentally persistent and have similarly been phased out in the U.S. for many years, though these two contaminants continue to often be found at levels of public health concern as discussed in section VI of this preamble. Currently, this also appears to be a similar trend for PFNA occurrence, where the drinking water sample data demonstrates it continues to occur at levels of public health concern. Therefore, in consideration of factors relating to the environmental persistence of PFNA, its presence in consumer products and possible continued use, and the observed occurrence trend of PFOA and PFOS, the EPA finds that there is a substantial likelihood PFNA occurs or will co-occur at a frequency and level of public health concern.

3. HFPO–DA

The occurrence data presented above, throughout section VI of this preamble, and discussed in the USEPA (2024b) support the agency's final determination that there is a substantial likelihood HFPO–DA occur with a frequency and at levels of public health concern in drinking water systems across the U.S. HFPO–DA was not included as a part of the UCMR 3; however, through analysis of available state data, which consisted of approximately 36,000 samples within approximately 10,100 systems, 10 of the 16 states that conducted non-targeted monitoring had state reported detections of HFPO–DA within 0.1 to 12.2 percent of their systems (Tables 2 and 3). These same states reported sample results ranging from 0.7 to 100 ng/L and median sample results ranging from 1.7 to 29.6 ng/L, demonstrating concentrations above the HRL of 10 ng/L.

Additionally, targeted state monitoring in North Carolina included sampling across six finished drinking water sites and 438 samples with HFPO–DA. Concentrations ranged from 9.52 to 1100 ng/L, a median concentration of 40 ng/L, and 433 (99 percent) samples exceeding the HRL (10 ng/L). When considering all available state data, there are at least 75 systems in 13 states serving more than 2.5 million people that reported any concentration of HFPO–DA, and at least 13 systems in 5 states within different geographic regions of the country serving a population of 227,000 people with reported concentrations above the HRL of 10 ng/L. Additionally, when evaluating only a subset of the available state data representing non-targeted monitoring to ensure that the data were not potentially over-represented by sampling completed in areas of known or suspected contamination, HFPO–DA was reported in approximately 0.48 percent of monitored systems; if these results were extrapolated to the nation and those system subject to the final rule requirements, the agency estimates that HFPO–DA would be detectable in over 320 PWSs serving 9.9 million people. If those results were further compared to the HRL for HFPO–DA (10 ng/L), HFPO–DA would be detected above the HRL in 42 systems with at least 495,000 people exposed. Finally, UCMR 5 data are being reported to the EPA while this final rule is being prepared. See section VI of this preamble for more information on the preliminary results. While these HFPO–DA UCMR 5 data are too preliminary to provide the basis for the regulatory determination, these preliminary UCMR 5 results appear to confirm the state data discussed above.

Further supporting this final determination, HFPO–DA is very stable and persistent in the environment. Additionally, unlike PFOA, PFOS, PFHxS, and PFNA which have been phased out in the U.S, HFPO–DA continues to be actively produced and used within the country and is generally considered to have replaced the production of PFOA. Since HFPO–DA is environmentally persistent and products containing HFPO–DA are still being actively produced and used, the EPA anticipates that contamination will continue, if not increase, due to disposal and breakdown in the environment. To illustrate this point further, PFOA and PFOS, two of the most extensively sampled PFAS, are also very environmentally persistent and have been phased out in the United States for many years, though these two PFAS continue to often be found at levels of public health concern as discussed in section VI of this preamble. Therefore, in consideration of factors relating to the environmental persistence of HFPO–DA, its continued and possibly increasing presence in consumer products and use, and the observed occurrence trend of PFOA and PFOS, the EPA anticipates that occurrence levels of HFPO–DA will similarly continue to be found at least to the levels described in this preamble demonstrating that there is a substantial likelihood HFPO–DA will occur at a frequency and level of public health concern.

As discussed, HFPO–DA continues to be actively produced and used throughout the U.S., it currently occurs at levels above its HRL, and it occurs within geographically diverse areas of the country demonstrating it is not a local or regional issue only. While the current individual occurrence profile of HFPO–DA is not as pervasive and is found at somewhat lower frequency as the currently observed levels of PFOA, PFOS, or PFHxS, based upon the available substantial amount of state occurrence data and given factors previously described, the EPA has determined that there is a substantial likelihood HFPO–DA occurs or will occur at a frequency and level of public health concern.

4. PFBS

The agency is deferring the final individual regulatory determination for PFBS to further consider whether occurrence information supports a finding that there is substantial likelihood that PFBS will individually occur in PWSs and at a level of public health concern. While current information demonstrates that PFBS frequently occurs, it has not been observed to exceed its HRL of 2,000 ng/L in isolation. However, when considered in mixture combinations with other PFAS, including PFHxS, PFNA, and HFPO–DA, PFBS is anticipated to have dose-additive adverse health effects (based on available data on PFAS and dose additivity) and there is a substantial likelihood of its co-occurrence in combinations of mixtures with PFHxS, PFNA, and HFPO–DA with a frequency and at levels of public health concern. This is described further in sections III.C.5 and VI.C. and VI.D of this preamble.

5. Mixtures of PFHxS, PFNA, HFPO–DA, and PFBS

Through the information presented within this section and in USEPA (2024b), along with the co-occurrence information presented in sections VI.C and VI.D of this preamble, the EPA's evaluation of all available UCMR 3 and state occurrence data demonstrates that there is a substantial likelihood that combinations of PFHxS, PFNA, HFPO–DA, and PFBS (collectively referred to as ''Hazard Index PFAS'') co-occur or will co-occur in mixtures at a frequency and level of public health concern.

As discussed throughout section III.C of this preamble, the EPA has determined that PFHxS, PFNA, and HFPO–DA each meet the second statutory criterion for individual regulation. Additionally, as demonstrated in sections VI.C. and D. of this preamble, the EPA has determined that these three PFAS also meet the second statutory criterion when present in mixture combinations. PFBS has not been observed to exceed its HRL of 2,000 ng/L in isolation; therefore, the EPA is deferring the individual regulatory determination for this PFAS to further consider future occurrence information. However, the agency has determined that PFBS frequently occurs (as shown in Table 2 and Table 3), and that when considering dose additivity there is a substantial likelihood of its co-occurrence in mixtures of PFHxS, PFNA, and/or HFPO–DA with a frequency and at a level of public health concern. Therefore, the agency has determined that PFBS also meets the criterion when present in mixture combinations with PFHxS, PFNA, and/or HFPO–DA.

In sections VI.C and D of this preamble, the EPA has presented its evaluation and findings related to the likelihood and frequency of co-occurrence of the four Hazard Index PFAS, including both through groupwise and pairwise analyses for the Hazard Index PFAS, in non-targeted state monitoring datasets. The groupwise co-occurrence analysis established the broad occurrence frequency of Hazard Index PFAS through a linkage to the presence of PFOA and PFOS. Because not as many states have monitored for the Hazard Index PFAS as compared to PFOA and PFOS, their occurrence information is less extensive than the occurrence information for PFOA and PFOS. Therefore, though the agency has previously made a final regulatory determination for PFOA and PFOS, establishing co-occurrence of Hazard Index PFAS with PFOA and PFOS is important to better understand the likelihood of Hazard Index PFAS occurrence. In this analysis, the six PFAS were separated into two groups—one consisted of PFOS and PFOA and the other group included the four Hazard Index PFAS. The analysis broke down the systems and samples according to whether chemicals from the two respective groups were detected. Given that the groupwise co-occurrence analysis established that there is a substantial likelihood that the Hazard Index PFAS frequently occur, particularly alongside PFOA or PFOS, the pairwise co-occurrence was relevant for understanding how the Hazard Index PFAS co-occur with each other instead of occurring independently. Pairwise co-occurrence analysis explored the odds ratios for each unique pair of PFAS included in the regulation. For every pair of PFAS chemicals included in the final regulation, the odds ratio, a statistic that, in this context, quantifies the strength of association between two PFAS being present, was found to be statistically significantly greater than 1. This means there was a statistically significant increase in the odds of reporting a chemical as present after knowing that the other chemical was detected. In most instances the odds appeared to increase in excess of a factor of ten. Thus, based on the large amount of available data, the chemicals are clearly demonstrated to frequently co-occur rather than occur independently of one another, supporting the agency's determination for mixtures of the four PFAS.

For the groupwise analysis, results generally indicated that when PFOA and PFOS were found, Hazard Index PFAS were considerably more likely to also be present. Additionally, for systems that only measured PFOA and/or PFOS and did not measure the Hazard Index PFAS, it can be assumed that the Hazard Index PFAS are more likely to be present in those systems, and that Hazard Index occurrence may be underestimated. Moreover, while PFOA and PFOS are not included within the Hazard Index PFAS or the determination to regulate mixtures of these PFAS, the pervasive occurrence of PFOA and PFOS shown in section VI of this preamble is a strong indicator that these other Hazard Index PFAS are also more likely to be found than what has been reported in state monitoring data to date. In this analysis, comparisons were also made between the number of Hazard Index PFAS analyzed and the number of Hazard Index PFAS reported present. As more Hazard Index PFAS were analyzed, more Hazard Index PFAS were reported present. Systems and samples where Hazard Index PFAS were found were more likely to find multiple Hazard Index PFAS than a single Hazard Index PFAS (when monitoring for three or four Hazard Index PFAS), demonstrating an increased likelihood of their co-occurrence. Additionally, for both system-level and sample-level analyses where PFOA and/or PFOS were reported present and all four Hazard Index PFAS were monitored, two or more Hazard Index PFAS were reported present more than half of the time, exhibiting they are more likely to occur together than in isolation. Furthermore, the EPA notes that when evaluating only a subset of the available state data representing non-targeted monitoring where either three or four Hazard Index PFAS were monitored, regardless of whether PFOA or PFOS were reported present, two or more of the Hazard Index PFAS were reported in approximately 12.1 percent of monitored systems; if these results were extrapolated to the nation, two or more of these four PFAS would co-occur in about 8,000 PWSs (see section VI.C.1 of this preamble for additional information).

The EPA uses a Hazard Index of 1 as the HRL to further evaluate the substantial likelihood of the Hazard Index PFAS co-occurring at a frequency and level of public health concern. As discussed in greater detail in section VI.D, of this preamble based on available state data the EPA finds that across 21 states there are at least 211 PWSs serving approximately 4.7 million people with results above a Hazard Index of 1 for mixtures including two or more of the Hazard Index PFAS. Specifically evaluating the presence of PFBS, in these same 211 systems where the Hazard Index was found to be greater than 1, PFBS was observed at or above its PQL in mixtures with one or more of the other three Hazard Index PFAS in at least 72 percent (152) of these systems serving approximately 4.5 million people. Additionally, as described previously in sections III.C.1–3, PFHxS, PFNA, HFPO–DA, and PFBS are all very stable and persistent in the environment. All are either still being actively used or legacy stocks may be used and imported into the U.S. Consequently, there is a substantial likelihood that environmental contamination of sources of drinking water from these PFAS will continue to co-occur to at least the levels described in this preamble.

Therefore, in consideration of the environmental persistence of these PFAS, their presence in consumer products and continued use, the findings of both the pairwise and groupwise co-occurrence analyses, and demonstration of combinations of Hazard Index PFAS mixtures exceeding the Hazard Index of 1, the EPA has determined there is sufficient occurrence information available to support the second criterion that there is a substantial likelihood that combinations of the four Hazard Index PFAS in mixtures co-occur at frequencies and levels of public health concern.

6. Summary of Major Public Comments and EPA Responses

The EPA requested comment on its preliminary regulatory determination for all four PFAS and their mixtures and its evaluation of the statutory criteria that supports the finding. The EPA also requested comment on additional occurrence data the agency should consider regarding its decision that PFHxS, PFNA, HFPO–DA, and PFBS and their mixtures occur or are substantially likely to occur in PWSs with a frequency and at levels of public health concern. The EPA received many comments on the agency's evaluation of the second statutory criterion under section 1412(b)(1)(A) of SDWA. Many commenters supported the EPA's preliminary determination that PFHxS, PFNA, HFPO–DA, and PFBS and mixtures of these four contaminants meet the second statutory occurrence criterion under SDWA.

A couple of commenters claimed that the EPA does not have a robust understanding of available occurrence data that supports any of the regulatory determinations for the four PFAS in this rule. Additionally, some commenters suggested that the preliminary determinations were ''rushed'' and ''non-scientific,'' and that the agency should wait until some or all of the UCMR 5 data is available and considered. The EPA disagrees. Sufficient occurrence data are available to establish a substantial likelihood of occurrence at frequencies and levels of health concern. Per the intent of the statute, the agency used the best available data in an expeditious manner, which, as the agency described earlier, was also a very large dataset consisting of tens of thousands of samples and representing one of the most robust occurrence datasets ever used to inform development of a drinking water regulation of a previously unregulated contaminant. The agency also disagrees that the occurrence analyses undertaken and available in the preamble as well as the technical support document for occurrence were non-scientific. Based on publicly available information within the state data, the EPA verified that the very large majority of samples (at least 97 percent) were collected using EPA-approved methods; the slight percentage the agency was unable to verify would not result in different agency conclusions. Additionally, the EPA notes that the aggregated data were assessed using precedented statistical metrics and analyses. In addition, the Cadwallader et al. (2022) model uses a robust, widely accepted Bayesian statistical approach for modeling contaminant occurrence. Based on these analyses, the EPA has a clear understanding of the occurrence of the modeled contaminants. As discussed in section III.C of this preamble and USEPA, 2024b, the EPA also has sufficient state data which consist of a greater number of total systems and samples than that included within the monitoring under UCMR 3, to confidently establish that there is a substantial likelihood of occurrence at frequencies and levels of public health concern.

As discussed above, the agency believes that the best currently available occurrence data demonstrate substantial likelihood of occurrence for the chemicals included in the final rule as they are demonstrated at frequencies and levels of public health concern. UCMR 5 data are being reported to the EPA while this final rule is being prepared. See section VI of this preamble for more information on the EPA's evaluation of the preliminary results. While these data are too preliminary to provide the basis for a regulatory determination, these preliminary UCMR 5 results appear to support the data discussed previously.

Several commenters disagreed that the available occurrence information supports a preliminary determination for HFPO–DA, with a few citing a lack of nationally representative data and suggesting a delay until UCMR 5 data is collected. The EPA disagrees with these comments, as the state monitoring data for the proposed rule demonstrates HFPO–DA occurrence in 13 geographically diverse states, including at 75 systems serving at least 2.5 million people. Moreover, non-national datasets may serve to demonstrate occurrence of a contaminant to warrant a positive determination and subsequent development of an NPDWR. For example, the best available HFPO–DA state data consists of approximately 36,000 samples within 10,000 systems and is representative of multiple geographic locations.

One commenter stated that a regulatory determination for PFNA was unnecessary as they do not believe it occurred with frequency under UCMR 3 monitoring, and a couple of other commenters suggested that a negative determination was appropriate for PFNA citing occurrence levels. The EPA disagrees that a negative determination is appropriate for PFNA as it has been demonstrated to occur at levels of public health concern in at least 52 water systems across 12 states. Furthermore, as described previously, when evaluating only a subset of the available state data representing non-targeted monitoring, PFNA was reported in approximately 3.6 percent of monitored systems and if those results were extrapolated across the country, PFNA would be detectable at any concentration in over 2,300 PWSs serving 21.2 million people and detectable above 10 ng/L in 227 systems serving 711,000 people. Additionally, PFNA frequently co-occurs with other PFAS, and as previously discussed in this section, presents dose additive health concerns with other PFAS demonstrating it is also an important component of the determination to regulate it in mixtures with PFHxS, HFPO–DA, and/or PFBS.

Commenters both agreed and disagreed with the EPA's individual preliminary determination for PFBS. With respect to commenters who suggested that the EPA has not met the occurrence criterion, while PFBS occurs at significant frequency, the agency is deferring the individual determination to regulate PFBS when it occurs individually until it conducts further evaluation under the statutory criteria. The EPA further finds that PFBS exposure may cause dose additive adverse health effects in mixtures with PFHxS, PFNA, and/or HFPO–DA; that there is a substantial likelihood that PFBS co-occurs in mixtures with PFHxS, PFNA, and/or HFPO–DA in PWSs with a frequency and at levels of public health concern; and that, in the sole judgment of the Administrator, regulation of PFBS in mixtures with PFHxS, PFNA, and/or HFPO–DA presents a meaningful opportunity for health risk reduction for persons served by PWSs. Therefore, PFBS will be regulated as part of a mixture with PFHxS, PFNA, and HFPO–DA.

A few commenters provided feedback on occurrence thresholds the agency should consider when evaluating the second statutory criterion for regulatory determinations. Particularly, these commenters recommended that the EPA should define a threshold for frequency and level of public health concern that warrants a specific regulatory determination. A few commenters cited other previous regulatory determinations where the agency made a determination not to regulate contaminants with similar or lower levels of occurrence suggesting that this should be the same for some or all of these four PFAS. Furthermore, some of these commenters stated that it would be arbitrary and capricious and conflict with the SDWA if the EPA did not use the level of adverse health effect (*i.e.,* the HRL) to represent the level at which a contaminant is considered a public health concern.

The EPA disagrees with these commenters and as demonstrated in the proposal and noted earlier in section III of this preamble, for this regulatory determination, as well as past determinations, the agency did compare available occurrence data relative to the contaminant HRL as a factor in informing the occurrence level of public health concern. However, the level of public health concern for purposes of the second criterion is a contaminant-specific analysis that include consideration of the HRL, as well as other factors and not solely based on the direct comparison to the HRL. There is not just one simple threshold used for public health concern for all contaminants. In the case of PFAS, this is particularly relevant given the dose-additivity of mixtures.

The EPA also disagrees with these commenters as SDWA does not define the occurrence level of public health concern for contaminants, nor does it prescribe the level of adverse health effects that must be used for a regulatory determination. Ultimately, the overall decision to regulate a contaminant considers all three statutory criteria, including the comprehensive assessment of meaningful opportunity which is in the Administrator's sole discretion. In previous EPA regulatory determinations, the agency has considered the occurrence criteria unique to the contaminant it is evaluating and has made decisions not to regulate contaminants both where there was substantial likelihood of occurrence at frequency and/or at levels of public health concern and where there was limited or no substantial likelihood of occurrence at frequency and/or at levels of public health concern. Consistent with this past regulatory history and the Administrator's authority under the terms of the statute, the decision considers all three criteria and cannot be determined in the exact same manner for different contaminants. While the EPA may have made negative determinations for other contaminants demonstrating occurrence at different frequencies and levels of public health concern, the basis for those decisions was specific to those contaminants and does not apply to these PFAS or any other future contaminants for which the EPA would make regulatory determinations. Therefore, the statute does not require, and the EPA does not use a minimum or one-size-fits-all occurrence thresholds (for either frequency or precise level) for regulatory determinations.

As described in section VI of this preamble, many commenters supported the EPA's proposal to regulate mixtures of PFAS. Specific to occurrence, some of these commenters particularly expressed support for the EPA's preliminary determination that mixtures of these four PFAS meet the second statutory occurrence criterion under SDWA, citing that the agency has used the best available information to determine that there is a substantial likelihood that combinations of these PFAS will co-occur in mixtures at a frequency and level of public health concern. One commenter stated that the additional occurrence data presented by the EPA in the proposal for the Hazard Index PFAS supports the EPA's proposed determination that these PFAS should be regulated under the SDWA. Conversely, several other commenters stated that there was not supporting evidence for the co-occurrence of the four Hazard Index PFAS. The EPA disagrees; the extent to which Hazard Index PFAS chemicals co-occur in the non-targeted state dataset is discussed extensively in the record for this rule and made evident through the system level analysis in section VI.C. of this preamble. As also discussed elsewhere in the record for this rule, in both system level and sample level analyses where PFOA and/or PFOS were reported present and all four Hazard Index PFAS were monitored, two or more Hazard Index PFAS were reported present more than half of the time. Further, the odds ratios tables in Exhibit 11 provide a statistical examination of pairwise co-occurrence. The odds ratio is a statistic that quantifies the strength of association between two events. In the context described here, an ''event'' is the reported presence of a specific PFAS contaminant. The odds ratio between PFOA and PFHxS, for example, reflects the strength of association between PFHxS being reported present and PFOA being reported present. If an odds ratio is greater than 1, the two events are associated. The higher the odds ratio, the stronger the association. For every pair of PFAS chemicals included in the proposed regulation, the odds ratio was found to be statistically significantly greater than 1. This means there was a statistically significant increase in the odds of a PFAS being present if the other PFAS compound was detected (*e.g.,* if PFOA is detected, PFHxS is more likely to also be found). In most instances the odds appeared to increase in excess of a factor of ten. Thus, based on the large amount of available data, the chemicals are clearly demonstrated to co-occur rather than occur independently of one another, further supporting the agency's determination for combinations of mixtures of the four PFAS.

After considering the public comments and additional occurrence data evaluated as requested by public commenters, the EPA finds that PFHxS, PFNA, and HFPO–DA individually and mixtures of these three PFAS and PFBS, meet the second statutory criterion for regulatory determinations under section 1412(b)(1)(A) of SDWA that the contaminant is known to occur or co-occur or there is a substantial likelihood that the contaminant will occur or co-occur in PWSs with a frequency and at levels of public health concern (USEPA, 2024b).

D. Statutory Criterion 3—Meaningful Opportunity

The agency has determined that individual regulation of PFHxS, PFNA, and HFPO–DA and regulation of combinations of PFHxS, PFNA, HFPO–DA, and PFBS in mixtures presents a meaningful opportunity for health risk reduction for persons served by PWSs. As discussed in section III.C. of this preamble, the EPA evaluated this third statutory criterion similarly to previous regulatory determinations using the Protocol developed under Regulatory Determination 3 (USEPA, 2014b) and also used in the Regulatory Determination 4. This evaluation includes a comprehensive assessment of meaningful opportunity for each unique contaminant including the nature of the health effects, sensitive populations affected, including infants, children and pregnant and nursing women, number of systems potentially affected, and populations exposed at levels of public health concern, geographic distribution of occurrence, technologies to treat and measure the contaminant, among other factors. The agency further reiterates that, per the statute, this determination of meaningful opportunity is in the Administrator's sole discretion.

Accordingly, the EPA is making this determination of meaningful opportunity after evaluating health, occurrence, treatment, and other related information and factors including consideration of the following:

- PFHxS, PFNA, and HFPO–DA and combinations of these three PFAS and PFBS in mixtures may cause multiple adverse human health effects, often at very low concentrations, on several biological systems including the endocrine, cardiovascular, developmental, renal, hematological, reproductive, immune, and hepatic systems as well as are likely to produce dose-additive effects from co-exposures.
- The substantial likelihood that PFHxS, PFNA, and HFPO–DA individually occur or will occur and that mixtures of PFHxS, PFNA, HFPO–DA, and/or PFBS co-occur or will co-occur together at frequencies and levels of public health concern in PWSs as discussed in section III of this preamble above and in section VI of this preamble, and the corresponding significant populations served by these water systems which potentially include sensitive populations and lifestages, such as pregnant and lactating women, as well as children.
- PFHxS, PFNA, HFPO–DA and combinations of these three PFAS and PFBS in mixtures are expected to be persistent in the environment, with some (*e.g.,* PFHxS, PFNA) also demonstrated to be very persistent in the human body.
- Validated EPA-approved measurement methods are available to measure PFHxS, PFNA, HFPO–DA, and

PFBS. See section VII of this preamble for further discussion.

- Treatment technologies are available to remove PFHxS, PFNA, and HFPO–DA and combinations of these three PFAS and PFBS from drinking water. See section X of this preamble for further discussion.
- Even though PFBS is very likely to be below its corresponding individual HRL when it occurs in a mixture, the record indicates that there is a substantial likelihood that it co-occurs with the regulated PFAS throughout public water systems nationwide. See sections III.C.5 and VI.C. of this preamble for further discussion. According to the 2023 Interagency PFAS Report to Congress (United States OSTP, 2023), PFBS has been shown to affect the following health endpoints: body weight, respiratory, cardiovascular, gastrointestinal, hematological, musculoskeletal, hepatic, renal, ocular, endocrine, immunological, neurological, reproductive, and developmental. Thus, including PFBS as a mixture component represents a meaningful opportunity to reduce PFBS' contributions to the overall hazard of the mixture and resulting dose additive health concerns. This is particularly relevant where the exposures of the other three PFAS in the mixture are also below their respective HRLs but when the hazard contributions of each mixture component are summed, the total exceeds the mixture HRL. In this scenario, the inclusion of PFBS allows for a more accurate picture of the overall hazard of the mixture so that PFBS can be reduced along with associated dose additive health concerns. In short, hazard would be underestimated if PFBS was not included in the regulated mixture. The EPA also considered the situation where PFHxS, PFNA, or HFPO–DA exceed one or more of their corresponding HRLs and co-occur with PFBS below its corresponding HRL. Although the exceedance of the mixture HRL is driven by a PFAS other than PFBS, PFBS is contributing to the overall hazard of the mixture and resulting dose additive health concerns. Including PFBS in the regulated mixture offers a meaningful opportunity to reduce dose additive health concerns because, when PFBS and other Hazard Index PFAS are present, public water systems will be able to better design and optimize their treatment systems to remove PFBS and any other co-occurring Hazard Index PFAS. This optimization will be even more effective knowing both that PFBS is present in source waters and its measured concentrations.
- Regulating PFHxS, PFNA, and HFPO–DA and combinations of these three PFAS and PFBS in mixtures is anticipated to reduce the overall public health risk from other PFAS, including PFOA and PFOS, that co-occur and are co-removed. Their regulation is anticipated to provide public health protection at the majority of known PWSs with PFAS-impacted drinking water.
- There are achievable steps to manage drinking water that can be taken to reduce risk.

As described in sections III.C, VI.C, VI.D, and USEPA (2024b), data from both the UCMR 3 and state monitoring efforts demonstrates the substantial likelihood of individual occurrence of PFHxS, PFNA, and HFPO–DA and co-occurrence of mixture combinations of PFHxS, PFNA, HFPO–DA, and PFBS at frequencies and levels of public health concern. Under UCMR 3, 5.7 million and 526,000 people had reported detections (greater than or equal to their minimum reporting levels which were two to three times their HRLs of 10 ng/L), of PFHxS and PFNA, respectively. Additionally, based on the more recent available state monitoring data presented earlier in this section, a range of geographically diverse states monitored systems that reported individual detections of PFHxS, PFNA, and HFPO–DA and serve approximate populations of 26.5 million, 2.5 million, and 8.4 million, respectively. Of these same systems, detections above the EPA's HRLs for PFHxS, PFNA, and HFPO–DA were seen in systems that serve approximate populations of 4.3 million, 227,000, and 177,000 people, respectively. As discussed previously, if these monitored systems were extrapolated to the nation, the EPA estimates that thousands of additional systems serving millions of people could have detectable levels of these three PFAS and hundreds of these systems may show values above the EPA's HRLs. Lastly, in evaluating the available state data, the EPA has found that mixtures of PFHxS, PFNA, HFPO–DA, and/or PFBS occur with a Hazard Index greater than 1 in systems serving approximately 4.7 million people. The agency further notes that while it has demonstrated through sufficient data that these four PFAS co-occur in mixtures at a frequency and level of public health concern in PWSs, throughout the nation it is extremely likely that additional systems and associated populations served would also demonstrate a Hazard Index greater than 1 if data for all PWSs were evaluated.

Analytical methods are available to measure PFHxS, PFNA, HFPO–DA, and PFBS in drinking water. The EPA has published two multi-laboratory validated drinking water methods for individually measuring PFHxS, PFNA, HFPO–DA, and PFBS. Additional discussion on analytical methods can be found in section VII of this preamble.

The EPA's analysis, summarized in section X of this preamble, found there are available treatment technologies capable of reducing PFHxS, PFNA, HFPO–DA, and PFBS. These technologies include granular activated carbon (GAC), anion exchange (AIX) resins, reverse osmosis (RO), and nanofiltration (NF). These treatment technologies remove PFHxS, PFNA, HFPO–DA, and PFBS and their mixtures. They also have been documented to co-remove other PFAS (Sörengård et al., 2020; McCleaf et al., 2017; Mastropietro et al., 2021). Furthermore, as described in section VI of this preamble, PFHxS, PFNA, HFPO–DA, and PFBS also co-occur with PFAS for which the agency is not currently making a regulatory determination. Many of these other emergent co-occurring PFAS are likely to also pose hazards to public health and the environment (Mahoney et al., 2022). Therefore, based on the EPA's findings that PFHxS, PFNA, HFPO–DA, and PFBS have a substantial likelihood to co-occur in drinking water with other PFAS and treating for PFHxS, PFNA, HFPO–DA, and PFBS is anticipated to result in removing these and other PFAS, individual regulation of PFHxS, PFNA, and HFPO–DA and regulation of mixtures of these three PFAS and PFBS also presents a meaningful opportunity to reduce the overall public health risk from all other PFAS that co-occur and are co-removed with PFHxS, PFNA, HFPO–DA, and PFBS.

With the ability to monitor for PFAS, identify contaminated drinking water sources and contaminated finished drinking water, and reduce PFAS exposure through management of drinking water, the EPA has identified meaningful and achievable actions that can be taken to reduce the human health risk of PFAS.

1. Proposal

The EPA made a preliminary determination that regulation of PFHxS, PFNA, HFPO–DA, and PFBS, both individually and in a mixture, presents a meaningful opportunity for health risk reduction for persons served by PWSs. The EPA made this preliminary determination after evaluating health, occurrence, treatment, and other related information against the three SDWA statutory criteria including consideration of the factors previously described in section III.D of this preamble above.

2. Summary of Major Public Comments and EPA Responses

The EPA received many comments on the agency's evaluation of the third statutory criterion under section 1412(b)(1)(A) of SDWA. Most commenters supported the EPA's evaluation under the preliminary determination that regulation of PFHxS, PFNA, HFPO–DA, PFBS and mixtures of these four contaminants presents a meaningful opportunity for health risk reduction and that the EPA had sufficiently justified this statutory criterion as well as the health and occurrence criterion. This included comments highlighting the extensive amount of work done by several states developing regulatory and non-regulatory levels for several PFAS compounds, including the PFAS for which the EPA is making regulatory determinations either individually or as a mixture. These commenters also noted the need for a consistent national standard for use in states where a state-specific standard has not yet been developed. Several commenters have also noted that although some states have developed or are in the process of developing their own state-level PFAS drinking water standards, regulatory standards currently vary across states. These commenters expressed concern that absence of a national drinking water standard has resulted in risk communication challenges with the public and disparities with PFAS exposure. Some commenters noted there are populations particularly sensitive or vulnerable to the health effects of these PFAS, including newborns, infants, and children. The EPA agrees with commenters that there is a need for a national PFAS drinking water regulation and that moving forward with a national-level regulation for PFHxS, PFNA, HFPO–DA, mixtures of these three PFAS and PFBS, as well as PFOA and PFOS, will provide improved national consistency in protecting public health and may reduce regulatory uncertainty for stakeholders across the country.

A few commenters expressed support for the EPA's evaluation of meaningful opportunity based on the treatment technologies which can remove the six PFAS for which the EPA is finalizing regulation. Furthermore, these commenters noted the meaningful opportunity to not only provide protection from the six regulated PFAS, but also other PFAS that will not be regulated as a part of this action.

Several commenters did not support the EPA's evaluation of the third statutory criterion, offering that in their opinion the EPA failed to justify that there is a meaningful opportunity for health risk reduction for the PFAS both individually and for their mixtures and stating that the EPA should consider other factors such as costs. A few of these commenters wrote that the EPA provided limited rationale and factors for its meaningful opportunity determination. The EPA disagrees with these commenters that the agency failed to justify that there is meaningful opportunity for health risk reduction or that the EPA provided limited rationale and factors in its meaningful opportunity evaluation for these contaminants individually and as mixtures. As described in the EPA's March 2023 proposal (USEPA, 2023f) and summarized previously, the EPA fully considered many factors both individually and within mixtures including individual contaminant and dose additive toxicity and health concerns, individual contaminant occurrence and co-occurrence of mixtures at frequencies and levels of public health concern, availability of similar treatment technologies to remove these four PFAS and analytical methods to measure them, and their individual and collective chemical and physical properties leading to their environmental persistence. Additionally, the EPA notes in this preamble, and as demonstrated through representative occurrence data, for the three contaminants individually and mixtures of the four, occurrence and co-occurrence is not only at a regional or local level, rather it covers multiple states throughout the country; therefore, a national level regulation is necessary to ensure all Americans served by PWSs are equally protected.

Some comments indicate that the health and occurrence information do not support that establishing drinking water standards presents a meaningful opportunity for health risk reduction. The agency disagrees with the commenters' assertion that the health and occurrence information are insufficient to justify a drinking water standard as supported in sections III.B. and III.C. of this preamble, and the agency finds that there is a meaningful opportunity for health risk reduction potential based upon multiple considerations including the population exposed to PFHxS, PFNA, HFPO–DA, and mixtures of these three PFAS and PFBS including sensitive populations and lifestages, such as newborns, infants and children.

Other comments assert that the EPA must evaluate the potential implementation challenges and cost considerations of regulation as part of the meaningful opportunity evaluation. The EPA disagrees with these commenters. The SDWA states that that the meaningful opportunity for overall health risk reduction for persons served by PWSs is in the sole judgement of the Administrator and does not require that the EPA consider costs for a regulatory determination. The SDWA does require that costs and benefits are presented and considered in the proposed rule's Health Risk Reduction Cost Analysis which the EPA did for the proposal and has updated as a part of the final rule within section XII.

A few other commenters provided that due to all of the additional human health exposure pathways other than drinking water for these PFAS, that regulation of drinking water would not represent a meaningful opportunity for overall health risk reduction. While the EPA recognizes that drinking water is one of several exposure routes, the EPA disagrees with these commenters. Removing the PFAS that have been found to occur or are substantially likely to occur from drinking water systems will result in a significant improvement in public health protection. The EPA also notes that through its *PFAS Strategic Roadmap* and associated actions, the agency is working expeditiously to address PFAS contamination in the environment and reduce human health PFAS exposure through all pathways. While beyond the scope of this rule, the EPA is making progress implementing many of the commitments in the Roadmap, including those that may significantly reduce PFAS source water concentrations.

E. The EPA's Final Determination Summary

The SDWA provides the EPA significant discretion when making a regulatory determination under section 1412(b)(1)(A). This decision to make a regulatory determination to individually regulate PFHxS, PFNA, and HFPO–DA and to regulate combinations of these three PFAS and PFBS in mixtures is based on consideration of the evidence supporting the factors individually and collectively.

The EPA's determination that PFHxS, PFNA, and HFPO–DA individually and mixtures of these three PFAS and PFBS "may have an adverse effect on the health of persons" is strongly supported by numerous studies. These studies demonstrate several adverse health effects, such as immune, thyroid, liver, kidney and developmental effects, and increased cholesterol levels, may occur following exposure to individual PFAS, and dose-additive health effects can occur following exposure to multiple PFAS at doses that likely would not individually result in these adverse health effects, but may pose health risks when combined in mixtures. Importantly, the best available peer reviewed science documents that these PFAS may have multiple adverse human health effects even at relatively low levels individually and when combined in mixtures (see section III.B.6.e f of this preamble or further information on studies supporting the conclusion of dose additivity).

The EPA's determination there is a substantial likelihood that the contaminant will occur in PWS with a frequency and at levels of public health concern is supported by evidence documenting the measured occurrence of PFHxS, PFNA, and HFPO–DA, and co-occurrence of these three PFAS and PFBS above the HRL, the stability and persistence of the contaminant in humans and/or the environment, and the current or legacy production and use in commerce.

Finally, the EPA's determination that individual regulation of PFHxS, PFNA, and HFPO–DA and regulation of these three PFAS and PFBS in mixtures presents a meaningful opportunity for health risks reductions is strongly supported by numerous factors, including the potential adverse human health effects at low levels and potential for exposure and co-exposure of these PFAS on sensitive populations and lifestages such as lactating and pregnant women and children, their persistence, and the availability of both analytical methods and treatment technologies to remove these contaminants in drinking water.

After considering these factors individually and together, the EPA has determined that PFHxS, PFNA, and HFPO–DA individually and mixtures of these three PFAS and PFBS meet the statutory criteria for regulation under SDWA. The EPA has an extensive record of information to make this determination now and recognizes the public health burden of these PFAS as well as PFOA and PFOS. The EPA notes the public urgency to reduce PFAS concentrations in drinking water described in the public comments. A PFAS NPDWR provides a mechanism to reduce these PFAS expeditiously for these impacted communities. In addition to making this final regulatory determination, the EPA is exercising its discretion to concurrently finalize MCLGs and NPDWRs for these PFAS as individual contaminants and for the specified PFAS mixtures in part to allow utilities to consider these PFAS specifically as they design systems to remove PFAS and to ensure that they are reducing these PFAS in their drinking water to the extent feasible and as quickly as practicable.

IV. MCLG Derivation

Section 1412(a)(3) of the Safe Drinking Water Act (SDWA) requires the Administrator of the Environmental Protection Agency (EPA) to publish a final MCLG simultaneously with the NPDWR. The MCLG is set, as defined in section 1412(b)(4)(A), at ''the level at which no known or anticipated adverse effects on the health of persons occur and which allows an adequate margin of safety.'' Consistent with SDWA section 1412(b)(3)(C)(i)(V), in developing the MCLG, the EPA considers ''the effects of the contaminant on the general population and on groups within the general population such as infants, children, pregnant women, the elderly, individuals with a history of serious illness, or other subpopulations that are identified as likely to be at greater risk of adverse health effects due to exposure to contaminants in drinking water than the general population.'' Other factors considered in determining MCLGs can include health effects data on drinking water contaminants and potential sources of exposure other than drinking water. MCLGs are not regulatory levels and are not enforceable. The statute does not dictate that the MCLG take a particular form; however, it must represent a ''level'' that meets the MCLG statutory definition. Given that the MCL must be ''as close as feasible'' to the MCLG, and that the MCL is defined as the ''maximum permissible level of a contaminant in water which is delivered to any user of a public water system,'' the MCLG can take any form so long as it is a maximum level of a contaminant in water.

Due to their widespread use and persistence, many PFAS are known to co-occur in drinking water and the environment—meaning that these contaminants are often together and in different combinations as mixtures (see sections III.C and VI of this preamble for additional discussion on occurrence). PFAS exposure can disrupt signaling of multiple biological pathways resulting in common adverse effects on several biological systems and functions, including thyroid hormone levels, lipid synthesis and metabolism, development, immune function, and liver function. Additionally, the EPA's examination of health effects information found that exposure through drinking water to a mixture of PFAS can act in a dose-additive manner (see sections III.B and IV.B of this preamble for additional discussion on mixture toxicity). Dose additivity means that exposure to multiple PFAS, at doses that individually would not be anticipated to result in adverse health effects, may pose health risks when combined in a mixture.

A. MCLG Derivation for PFOA and PFOS

To establish an MCLG for individual contaminants, the EPA assesses the peer-reviewed science examining cancer and noncancer health effects associated with oral exposure to the contaminant. For known or likely linear carcinogenic contaminants, where there is a proportional relationship between dose and carcinogenicity at low concentrations or where there is insufficient information to determine that a carcinogen has a threshold dose below which no carcinogenic effects have been observed, the EPA has a long-standing practice of establishing the MCLG at zero (see USEPA, 1998a; USEPA, 2000c; USEPA, 2001; *See* S. Rep. No. 169, 104th Cong., 1st Sess. (1995) at 3). For nonlinear carcinogenic contaminants, contaminants that are designated as *Suggestive Human Carcinogens* (USEPA, 2005a), and non-carcinogenic contaminants, the EPA typically establishes the MCLG based on a noncancer RfD. An RfD is an estimate of a daily oral exposure to the human population (including sensitive populations) that is likely to be without an appreciable risk of deleterious effects during a lifetime. A nonlinear carcinogen is a chemical agent for which the associated cancer response does not increase in direct proportion to the exposure level and for which there is scientific evidence demonstrating a threshold level of exposure below which there is no appreciable cancer risk.

1. Proposal

To support the proposed rule, the EPA published PFOA and PFOS draft toxicity assessments and the proposed MCLGs for public comment (USEPA, 2023g; USEPA, 2023h). Prior to conducting the systematic review for the PFOA and PFOS draft toxicity assessments, the EPA established the internal protocols for the systematic review steps of literature search, Population, Exposure, Comparator, and Outcomes (PECO) development, literature screen, and study quality evaluation. The EPA incorporated detailed, transparent, and complete protocols for all steps of the systematic review process (USEPA, 2023g; USEPA, 2023h; USEPA, 2023i; USEPA, 2023j). Additionally, the EPA updated and expanded the protocols and methods based on SAB recommendations to improve the transparency of the process the EPA used to derive the MCLGs for PFOA and PFOS and to improve consistency with the *ORD Staff Handbook for Developing IRIS Assessments* (USEPA, 2022f). The EPA followed this transparent systematic review process to evaluate the best available peer-reviewed science and to determine the weight of evidence for carcinogenicity and the cancer classifications for PFOA and PFOS according to agency guidance (USEPA, 2005a).

Based on the EPA's analysis of the best available data and following agency guidance, the EPA determined that both PFOA and PFOS are *Likely to be Carcinogenic to Humans* based on sufficient evidence of carcinogenicity in humans and animals (USEPA, 2005a; USEPA, 2023g; USEPA, 2023h). The EPA also determined that a linear default extrapolation approach is appropriate for PFOA and PFOS as there is no evidence demonstrating a threshold level of exposure below which there is no appreciable cancer risk for either compound (USEPA, 2005a). Therefore, the EPA concluded that there is no known threshold for carcinogenicity. Based upon a consideration of the best available peer-reviewed science and the statute's directive that the MCLG be ''set at the level at which no known or anticipated adverse effects on the health of persons occur and which allow an adequate margin of safety,'' the EPA proposed MCLGs of zero for both PFOA and PFOS in drinking water. Setting the MCLG at zero under these conditions is also supported by long standing practice at the EPA's Office of Water for *Likely* or *Known Human Carcinogens* (see USEPA, 1998a; USEPA, 2000c; USEPA, 2001; USEPA, 2016b; *See* S. Rep. No. 169, 104th Cong., 1st Sess. (1995) at 3).

2. Summary of Major Public Comments and EPA Responses

The EPA requested comment on both the toxicity assessment conclusions and the proposed MCLG derivation for PFOA and PFOS. In this section the EPA focuses the summary of public comments and responses on comments related to the cancer classification determinations for PFOA and PFOS because that was the basis for the proposed MCLG derivations (USEPA, 2023g; USEPA, 2023h). The noncancer health effects that the EPA identified as hazards in the draft toxicity assessments (*i.e.,* decreased immune response in children, increased alanine aminotransferase (ALT), decreased birth weight and increased cholesterol) were not the basis for the proposed MCLG derivation. Importantly, an MCLG of zero is also protective of noncancer endpoints which were evaluated in the EPA's HRRCA (Health Risk Reduction and Cost Analysis). Comments related to the benefits the EPA quantified that are associated with noncancer health effects are described in section XII.

A few commenters agreed with the systematic review protocol the EPA used to evaluate the studies that supported the PFOA and PFOS cancer classification determinations in the draft toxicity assessments (USEPA, 2023g; USEPA, 2023h; USEPA, 2023i; USEPA, 2023j), with one commenter stating that the approach was ''thorough and well-reasoned.'' Commenters stated that the systematic review protocol was clear because the EPA had addressed all concerns highlighted during the peer review process.

One commenter stated that the EPA did not conduct a systematic review of the literature and did not follow the *ORD Staff Handbook for Developing IRIS Assessments* (USEPA, 2022f) to develop the toxicity assessments for PFOA and PFOS. This commenter stated the EPA lacked ''a predefined protocol'' and that the ''systematic review methods lack[ed] transparency and consistency.'' The commenter took particular issue with the EPA's protocols for study quality evaluations, stating that they were inconsistent and not aligned with the *ORD Staff Handbook for Developing IRIS Assessments* (USEPA, 2022f). The EPA disagrees with this commenter's claims. The EPA adopted the overall approach and steps in the *ORD Staff Handbook for Developing IRIS Assessments* (USEPA, 2022f) and the *Systematic Review Protocol for the PFAS IRIS Assessments* (USEPA, 2021h) to develop PFOA- and PFOS-specific protocols that then formed the basis for performing study quality evaluations, evidence integration, and critical study selection (see appendix A in USEPA, 2023g; USEPA, 2023h; USEPA, 2023i; USEPA, 2023j). This predefined protocol was made available for public comment as appendix A of the toxicity assessments (USEPA, 2023i; USEPA, 2023j). Importantly, the EPA's Office of Water collaborated with the EPA's Office of Research and Development in conducting study quality evaluations, evidence integration, and selection of critical studies to ensure consistency with the *ORD Staff Handbook for Developing IRIS Assessments* (USEPA, 2022f) and the *Systematic Review Protocol for the PFAS IRIS Assessments* (USEPA, 2021h).

A few commenters claimed that the EPA did not use the best available science when developing the toxicity assessments for PFOA and PFOS, asserting that the EPA did not follow its own guidance or data quality standards and that the EPA's systematic review process was flawed (see discussion above). The EPA disagrees with these commenters' claims. The EPA has followed statutory requirements to use the best available peer-reviewed science in two respects: by (1) considering relevant peer-reviewed literature identified by performing systematic searches of the scientific literature or identified through public comment and (2) relying on peer-reviewed, published EPA human health risk assessment methodology as well as systematic review best practices (USEPA, 2021h; USEPA, 2022f). The risk assessment guidance and best practices serve as the basis for the PFOA and PFOS health effects systematic review methods used to identify, evaluate, and quantify the available data. Not only did the EPA incorporate literature identified in previous assessments, as recommended by the SAB (USEPA, 2022i), but the EPA also conducted several updated systematic literature searches, the most recent of which was completed in February 2023. This approach ensured that the literature under review encompassed studies included in the 2016 *Health Effects Support Documents* (HESDs) (USEPA, 2016c; USEPA, 2016d) and recently available studies. The results of the most recent literature search provide further support for the conclusions made in the draft toxicity assessments for PFOA and PFOS (USEPA, 2023g; USEPA, 2023h) and are described in appendix A of the final toxicity assessments (USEPA, 2024h; USEPA, 2024i).

As described above, the PFOA and PFOS systematic review protocol is consistent with the *ORD Staff Handbook for Developing IRIS Assessments* (USEPA, 2022f) and also considers PFOA- and PFOS-specific protocol updates outlined in the *Systematic Review Protocol for the PFBA, PFHxA, PFHxS, PFNA, and PFDA (anionic and acid forms) IRIS Assessments* (USEPA, 2021h). The EPA additionally followed human health risk assessment methods for developing toxicity values (*e.g.,* USEPA, 2002a), conducting benchmark dose (BMD) modeling (USEPA, 2012), and other analyses. In the PFOA and PFOS toxicity assessments and the appendices, the EPA clearly describes the methods used and how those methods and decisions are consistent with the EPA practices and recommendations (*i.e.,* through quotes and citations) described in various guidance documents.

One commenter stated that the EPA did not use the best available peer-reviewed science because the assessments did not follow methodological or statistical guidance. Specifically, this commenter stated the EPA did not follow *A Review of the Reference Dose and Reference Concentration Processes* (USEPA, 2002a) when selecting uncertainty factors and claimed the EPA did not follow guidance on data quality (USEPA, 2003; USEPA, 2006b; USEPA, 2014b). The commenter stated they believed the assessments contained flaws including exclusion of covariates in modeling, reliance on peer-reviewed studies published by non-EPA employees, and an inability to replicate results. The EPA disagrees with these comments. Regarding data quality control, data quality objectives are an integral part of the *ORD Staff Handbook for Developing IRIS Assessments* (USEPA, 2022f) and many of the concepts outlined in data quality guidance recommended by the commenter (USEPA, 2003; USEPA, 2006b; USEPA, 2014b) are addressed through the EPA's use of the *ORD Handbook* (USEPA, 2022f). Furthermore, this work was conducted under a programmatic quality assurance project plan (QAPP) which ensures that all EPA data quality guidance is followed, including those cited by the commenter. Additionally, by developing and implementing a systematic review protocol consistent with the *ORD Handbook* (USEPA, 2022f), the EPA reduced potential confirmation bias, a concern raised by another commenter, by conducting multiple independent evaluations of studies, relying on a data-driven, weight of evidence approach, and by incorporating expertise from across the agency.

In many cases the commenters have misinterpreted the methods and decisions the EPA used to analyze the data or misinterpreted the guidance itself. For example, one commenter mistakenly suggested that the EPA did not consider covariates in its analyses of epidemiological studies; the EPA described which covariates were considered in each analysis in several sections of the draft toxicity assessments and appendices (USEPA, 2023g; USEPA, 2023h; USEPA, 2023i; USEPA, 2023j), including in descriptions of the studies in section 3 and modeling of the studies in appendix E. The EPA also notes that the primary studies that provide the data describe covariate adjustments in their published analyses.

A couple of commenters suggested that the toxicity assessments for PFOA and PFOS were not adequately peer-reviewed because changes were made post peer review (*i.e.,* after publication of the final report by the SAB PFAS Review Panel (USEPA, 2022i)), the most significant of which was the updated cancer classification for PFOS, but also included the addition of figures and mechanistic syntheses. The EPA disagrees with this assertion. The toxicity assessments, including the conclusions that are material to the derivation of the MCLGs, were peer-reviewed by the SAB PFAS review panel (USEPA, 2022i). Notably, this panel ''agreed with many of the conclusions presented in the assessments, framework and analysis'' (USEPA, 2022i). The only assessment conclusion that changed and impacted MCLG derivation between SAB review and rule proposal was that the cancer classification for PFOS of *Suggestive Evidence of Carcinogenicity* was updated to *Likely to be Carcinogenic to Humans* according to the *Guidelines for Carcinogen Risk Assessment* (USEPA, 2005a). This conclusion for PFOS was based on a reevaluation of the available data in response to multiple comments from the SAB PFAS review panel stating that ''[s]everal new studies have been published that warrant further evaluation to determine whether the 'likely' designation is appropriate'' for PFOS and that the EPA's ''interpretation of the hepatocellular carcinoma data from the Butenhoff et al. (2012) study in the 2016 HESD is overly conservative in dismissing the appearance of a dose-response relationship for this endpoint, particularly in females'' (USEPA, 2022i). In responding to the SAB's recommendation that the EPA provide an ''explicit description of why the available data for PFOS do not meet the EPA *Guidelines for Carcinogen Risk Assessment* (2005) criterion for the higher designation as 'likely carcinogenic,' '' and taking into consideration recently published peer-reviewed epidemiological studies demonstrating concordance in humans identified through the final updated literature search recommended by the SAB, the EPA determined that PFOS meets the criterion for the higher designation of *Likely to Be Carcinogenic to Humans* (USEPA, 2005a). This decision was described in sections 3.5.5 and 6.4 of the draft assessment (USEPA, 2023h). Additional discussion regarding the PFOS cancer descriptor decision is provided here.

One commenter stated that the EPA addressed the SAB's concerns regarding the systematic review protocol in the documents supporting the proposed rulemaking. A few commenters reiterated the importance of the SAB's recommendations, including to more thoroughly describe systematic review methods used in the assessment (*e.g.,* study inclusion and exclusion criteria), incorporate additional epidemiological studies, provide rationale for critical study selection, and derive candidate toxicity values from both human and animal data. In contrast, a few commenters claimed that the EPA did not adequately consider several recommendations made by the SAB PFAS Review Panel in their final report (USEPA, 2022i), including that the EPA did not incorporate studies from the 2016 HESDs (USEPA, 2016c; USEPA, 2016d) or develop multiple cancer slope factors (CSFs). One commenter requested clarification on whether the EPA had implemented the feedback from the SAB.

The EPA disagrees with the comments that the agency did not ''meaningfully implement'' SAB feedback. The EPA agrees with commenters that highlighted the importance of the SAB's suggestions, and notes that the EPA addressed the SAB's recommendations to more thoroughly explain the systematic review protocol and expand the systematic review protocol beyond study quality evaluation and data extraction in the draft toxicity assessments published at the time of rule proposal (USEPA, 2023g; USEPA, 2023h; USEPA, 2023i; USEPA, 2023j). As outlined in the *EPA Response to Final Science Advisory Board Recommendations (August 2022) on Four Draft Support Documents for the EPA's Proposed PFAS National Primary Drinking Water Regulation* (USEPA, 2023k), the EPA considered all of the comments and recommendations from the SAB and made substantial improvements to address the reported concerns prior to publishing the public comment draft assessments (USEPA, 2023g; USEPA, 2023h). The EPA published a response to SAB comments document that detailed how the agency considered and responded to the SAB PFAS Review Panel's comments at the time of rule proposal (USEPA, 2023k). The resulting draft toxicity assessments and protocol released for public comment along with the proposed rule reflect improvements including thorough and detailed descriptions of the methods used during assessment development, inclusion of epidemiological studies from the 2016 HESDs for PFOA and PFOS in the systematic review (USEPA, 2016c; USEPA, 2016d), updates to the literature, implementation of an evidence integration framework, expansion of rationale for critical study and model selections, development of toxicity values from both animal toxicological and epidemiological data, when warranted, and many other actions. The EPA appreciated the SAB's engagement, extensive review, and comments on the *Proposed Approaches* documents (USEPA, 2021i; USEPA, 2021j). Furthermore, the EPA provided its consideration of every recommendation the SAB provided when updating and finalizing the assessments for PFOA and PFOS at the time of rule proposal (USEPA, 2023k).

Many commenters agreed that that available data indicate that exposure to either PFOA or PFOS is associated with cancer in humans and supported the EPA's determination that PFOA and PFOS are *Likely to be Carcinogenic to Humans* according to the *Guidelines for Carcinogen Risk Assessment* (USEPA, 2005a). Multiple commenters agreed that studies published since the 2016 HESDs (USEPA, 2016c; USEPA, 2016d) have strengthened this conclusion. In particular, one commenter supported the EPA's conclusions regarding the human relevance of hepatic and pancreatic tumors observed in rats administered PFOS, citing their own independent health assessment conclusion that ''several lines of evidence do not support a conclusion that liver effects due to PFOS exposure are PPARα-dependent'' and therefore, may be relevant to humans (NJDWQI, 2018).

Several commenters disagreed with the EPA's determinations that PFOA and PFOS are each *Likely to be Carcinogenic to Humans.* Two commenters claimed that the tumor types observed in rats (*e.g.,* hepatic tumors) after PFOA or PFOS administration are not relevant to humans. Some commenters also stated that the human data do not support an association between PFOS exposure and cancer. One commenter specifically claimed that Shearer et al. (2021) does not provide sufficient evidence for changing PFOS's cancer classification from *Suggestive Evidence of Carcinogenicity* to *Likely to be Carcinogenic to Humans* because it did not report associations between PFOS exposure and risk of renal cell carcinoma (RCC). Two commenters stated that the EPA's discussion using structural similarities between PFOA and PFOS to support evidence of the carcinogenicity of PFOS was inconsistent with the *Guidelines for Carcinogen Risk Assessment* (USEPA, 2005a). A few commenters additionally questioned or disagreed with the determination that PFOA is *Likely to be Carcinogenic to Humans* because of uncertainties in the epidemiological database and a lack of evidence indicating that PFOA is genotoxic.

The EPA disagrees with these comments. With respect to the human relevance of the animal tumors observed in rats after chronic oral exposure to either PFOA or PFOS, the EPA considered all hypothesized modes of action (MOAs) and underlying carcinogenic mechanisms in its cancer assessments, including those that some commenters have argued are irrelevant to humans (*e.g.,* peroxisome proliferator-activated receptor α (PPARα) activation), the discussion for which is available in section 3.5.4.2 of the toxicity assessments for PFOA and PFOS (USEPA, 2024c; USEPA, 2024d). After review of the available mechanistic literature for PFOA and PFOS, the EPA concluded that there are multiple plausible mechanisms, including some that are independent of PPARα, that may contribute to the observed carcinogenicity of either PFOA or PFOS in rats. Further confirmatory support for the EPA's conclusions regarding multiple plausible mechanisms of carcinogenicity comes from literature reviews published by state and global health agencies which concluded that the liver tumors associated with PFOA and/or PFOS exposure may not entirely depend on PPARα activation and therefore may be relevant to humans (CalEPA, 2021; IARC, 2016; NJDWQI, 2017; NJDWQI, 2018).

Additionally, the EPA did not rely on results reported by Shearer et al. (2021) as a rationale for updating the cancer classification for PFOS to *Likely to be Carcinogenic to Humans* (USEPA, 2005a) and acknowledges uncertainties in the results from this study, including that the effect in the third PFOS exposure quartile was null, the effects were attenuated (*i.e.,* reduced in magnitude) when adjusted for exposure to other PFAS, and there was no association when exposure to PFOS was considered as a continuous variable, rather than when PFOS exposure levels were stratified by quartiles (USEPA, 2023h). As described in sections 3.5.5 and 6.4 of the draft PFOS toxicity assessment, the available information exceeds the characteristics for the classification of *Suggestive Evidence of Carcinogenic Potential* (USEPA, 2005a) because there is statistically significant evidence of multi-sex and multi-site tumorigenesis from a *high* confidence animal toxicological study, as well as mixed but plausible evidence of carcinogenicity in humans and mechanistic data showing potential human relevance of the observed tumor data in animals (USEPA, 2023h). The EPA notes that the recently published studies reporting associations between PFOS exposure and hepatocellular carcinoma in humans (Goodrich et al., 2022; Cao et al., 2022) further strengthen the epidemiological database and support the cancer classification of *Likely to be Carcinogenic to Humans* for PFOS.

Regarding commenters' claims that the EPA used the structural similarities between PFOA and PFOS as supporting evidence of the carcinogenic potential of PFOS, the EPA did not rely on structural similarities to draw conclusions about the cancer classification (see rationale listed above) but instead used this information as supplemental support for the *Likely* classification. The EPA originally included this supplemental line of evidence because the *Guidelines for Carcinogen Risk Assessment* (USEPA, 2005a) explicitly states that ''[a]nalogue effects are instructive in investigating carcinogenic potential of an agent as well as in identifying potential target organs, exposures associated with effects, and potential functional class effects or modes of action.'' PFOA and PFOS differ in their chemical structure by a single functional group; nevertheless, since a full structure-activity relationship analysis was not conducted, the EPA removed discussion on this supplemental line of evidence from the final toxicity assessment for PFOS (USEPA, 2024d).

Further, the EPA disagrees with comments stating that the epidemiological database for PFOA is too uncertain to support a classification of *Likely to be Carcinogenic to Humans* (USEPA, 2005a). As described in both the draft (USEPA, 2023g) and final toxicity assessments for PFOA (USEPA, 2024c), as well as the *Maximum Contaminant Level Goals for Perfluorooctanoic Acid (PFOA) and Perfluorooctane Sulfonic Acid (PFOS)* document (USEPA, 2024j) the available data support an increased risk of both kidney and testicular cancers associated with PFOA exposure. There is also evidence that PFOA exposure may be associated with an increased breast cancer risk, based on studies in populations with specific polymorphisms and for specific types of breast tumors. Taken together, these results provide consistent and plausible evidence of PFOA carcinogenicity in humans. Additionally, the EPA notes that while genotoxicity is one potential MOA leading to carcinogenicity, there is no requirement that a chemical be genotoxic for the EPA to classify it as either *Carcinogenic to Humans, Likely to be Carcinogenic to Humans,* or *Suggestive Evidence of Carcinogenic Potential* according to the *Guidelines for Carcinogen Risk Assessment* (USEPA, 2005a). Importantly, the SAB PFAS Review Panel supported the *Likely to be Carcinogenic to Humans* designation for PFOA in its final report (USEPA, 2022i).

Many commenters supported the EPA's proposed MCLGs of zero for both PFOA and PFOS, citing well-documented health effects, including cancer, resulting from exposure to either PFOA or PFOS as rationale for their support of the proposed rulemaking. Several commenters also agreed with the EPA's long-standing practice of establishing the MCLG at zero (see USEPA, 1998a; USEPA, 2000c; USEPA, 2001; *See* S. Rep. No. 169, 104th Cong., 1st Sess. (1995) at 3) for known or likely linear carcinogenic contaminants, with one commenter stating that it is ''appropriate based on the weight of evidence for carcinogenicity and other adverse health impacts of PFOA and PFOS at very low exposures.''

Two commenters disagreed with MCLGs of zero for PFOA and PFOS, with one commenter claiming that the EPA's determinations were ''not consistent with the evidence the EPA presents nor with its own guidance'' (*i.e.,* the EPA's cancer assessment was not consistent with assessment approaches recommended in the *Guidelines for Carcinogen Risk Assessment* (USEPA, 2005a)). The EPA disagrees with these commenters' assertions because there is sufficient weight of evidence for carcinogenic risk of both PFOA and PFOS exposures supporting a classification of *Likely to be Carcinogenic to Humans* according to the *Guidelines for Carcinogen Risk Assessment* (USEPA, 2005a) from the available epidemiological and animal toxicological studies. Consistent with the guidelines, the EPA provided a narrative to ''explain the case for choosing one descriptor and discuss the arguments for considering but not choosing another'' (USEPA, 2005a) in the draft and final toxicity assessments (USEPA, 2024c; USEPA, 2024d; USEPA, 2023g; USEPA, 2023h).

3. Final Rule

Based on the best available peer-reviewed science and consistent with agency guidance (USEPA, 2005a), the EPA has determined that both PFOA and PFOS are *Likely to be Carcinogenic to Humans.* Therefore, following established agency practice regarding contaminants with this classification and consistent with the statutory directive to set an MCLG ''at the level at which no known or anticipated adverse effects on the health of persons occur and which allows for an adequate margin of safety,'' the EPA set individual MCLGs for both PFOA and PFOS at zero. As described above, the EPA used the best available peer-reviewed science, followed agency guidance and current human health risk assessment methodology, including the *ORD Staff Handbook for Developing IRIS Assessments* (USEPA, 2022f) and the *Guidelines for Carcinogen Risk Assessment* (USEPA, 2005a), and adequately peer-reviewed (USEPA, 2022i) the science underlying the MCLG derivation for both PFOA and PFOS (USEPA, 2024c; USEPA, 2024d; USEPA, 2024j).

Consistent with the *Guidelines for Carcinogen Risk Assessment* (USEPA, 2005a), the EPA reviewed the weight of evidence and determined that PFOA and PFOS are each designated as *Likely to Be Carcinogenic to Humans,* because ''the evidence is adequate to demonstrate carcinogenic potential to humans but does not reach the weight of evidence for the descriptor Carcinogenic to Humans.'' For PFOA, this determination was based on the evidence of kidney and testicular cancer in humans and Leydig cell tumors, pancreatic acinar cell tumors, and hepatocellular tumors in rats as described in USEPA (2024c). For PFOS, this determination was based on the evidence of hepatocellular tumors in male and female rats, which is further supported by recent evidence of hepatocellular carcinoma in humans (Goodrich et al., 2022; Cao et al., 2022), pancreatic islet cell carcinomas in male rats, and mixed but plausible evidence of bladder, prostate, kidney, and breast cancers in humans (USEPA, 2024d). The EPA has updated and finalized the toxicity assessment for PFOS to reflect the new epidemiological evidence (USEPA, 2024d; USEPA, 2024i).

Consistent with the statutory definition of MCLG, the EPA establishes MCLGs of zero for carcinogens classified as either *Carcinogenic to Humans* or *Likely to be Carcinogenic to Humans* where there is a proportional relationship between dose and carcinogenicity at low concentrations or where there is insufficient information to determine that a carcinogen has a threshold dose below which no carcinogenic effects have been observed. In these situations, the EPA takes the health protective approach of assuming that carcinogenic effects should therefore be extrapolated linearly to zero. This is called the linear default extrapolation approach. This approach ensures that the MCLG is set at a level where there are no known or anticipated adverse health effects, allowing for an adequate margin of safety. Here, the EPA has determined that PFOA and PFOS are *Likely to be Carcinogenic to Humans* based on sufficient evidence of carcinogenicity in humans and animals (USEPA, 2024c; USEPA, 2024d). The EPA has also determined that a linear default extrapolation approach is appropriate as there is no evidence demonstrating a threshold level of exposure below which there is no appreciable cancer risk (USEPA, 2005a). Based on this lack of evidence, the EPA concluded that there is no known threshold for carcinogenicity. Based upon a consideration of the best available peer-reviewed science and statutory directive to set the MCLG ''at the level at which no known or anticipated adverse effects on the health of persons occur and which allows an adequate margin of safety,'' the EPA has finalized MCLGs of zero for PFOA and PFOS in drinking water.

While not a basis for the EPA's MCLG, the EPA notes that its toxicity assessments indicate either PFOA or PFOS exposure are also associated with multiple non-cancer adverse health effects. The PFOA and PFOS candidate non-cancer RfDs based on human epidemiology studies for various health outcomes (*i.e.,* developmental, cardiovascular, immune, and hepatic) range from 2×10^{-7} to 3×10^{-8} mg/kg/day (USEPA, 2024c; USEPA, 2024d; USEPA, 2024h; USEPA, 2024i).

B. MCLG Derivation for Additional PFAS

Section 1412(b)(4)(A) requires the EPA to set the MCLG at a ''level at which no known or anticipated adverse effects on the health of persons occur and which allows an adequate margin of safety.'' In this action, the EPA is setting MCLGs (and MCLs) for five individual PFAS (section IV.C of this preamble) as well as for mixtures of three of these PFAS plus PFBS. In the context of this NPDWR, the Hazard Index is a method which determines when a mixture of two or more of four PFAS—PFHxS, PFNA, HFPO–DA, and PFBS—exceeds the level of health concern with a margin of safety and thus the Hazard Index (equal to 1) is the MCLG for any mixture of those four PFAS. Based on the scientific record, each PFAS within the mixture has a HBWC, which is set at the level below which adverse effects are not likely to occur and allows for an adequate a margin of safety. See USEPA, 2024f and section IV.B. of this preamble. The scientific record also shows that PFHxS, PFNA, HFPO–DA, and PFBS elicit the same or similar profiles of adverse health effects in several biological organs and systems, but with differing potencies for effect(s) (see USEPA, 2022i and 2024a; and section IV.B of this preamble). As a result, as discussed elsewhere in the preamble, PFAS that elicit similar observed adverse health effects following individual exposure should be assumed to act in a dose-additive manner when in a mixture unless data demonstrate otherwise (USEPA, 2024a). See USEPA, 2024a and section II and IV.B of this preamble. This means that where drinking water contains any combination of two or more of these PFAS, the hazard associated with each PFAS in the mixture must be added up to determine whether the mixture exceeds a level of public health concern.

The Hazard Index is the method for calculating this level (*i.e.,* the mixture MCLG) and reflects both the measured amount of each of the four PFAS in the mixture and the toxicity (represented by the HBWC) of each of the four PFAS. The PFAS mixture Hazard Index is an approach to determine whether any mixture of two or more of these four PFAS in drinking water exceeds a level of health concern by first calculating the ratio of the measured concentration of each of the four PFAS divided by its toxicity (the HBWC). This results in the ''hazard quotient'' (HQ) for each of the four PFAS. Because the health effects of these PFAS present dose additive concerns (USEPA, 2024a), the four HQs are added together, and if the result exceeds 1, then the hazard from the combined amounts of the four PFAS in drinking water exceeds a level of public health concern.

1. MCLG Derivation for a PFAS Mixture

a. Proposal

The EPA proposed a Hazard Index MCLG to protect public health from exposure to mixtures of any combination of PFHxS, PFNA, HFPO–DA, and/or PFBS, four PFAS that elicit a shared set of adverse effects and co-occur in drinking water. The Hazard Index is an approach based on dose additivity that has been validated and used by the EPA to assess chemical mixtures in several contexts (USEPA, 1986; USEPA, 2000a; USEPA, 2022i). The EPA's proposal was based on the agency's finding that the Hazard Index approach is the most practical approach for establishing an MCLG for PFAS mixtures that meets the statutory requirements outlined in section 1412(b)(1)(A) of SDWA. This is because the Hazard Index assesses the exposure level of each component PFAS relative to its HBWC, which is based on the most sensitive known adverse health effect (based on the weight of evidence) and considers sensitive population(s) and life stage(s) as well as potential exposure sources beyond drinking water. Furthermore, the Hazard Index accounts for dose additive health concerns by summing the hazard contribution from each mixture component to ensure that the mixture is not exceeding the level below which there are no known or anticipated adverse health effects and allows for an adequate margin of safety.

The proposal defined a mixture as containing one or more of the four PFAS and therefore covered each contaminant individually if only one of the four PFAS occurred. Thus, the Hazard Index as proposed ensures that the level of exposure to an individual PFAS remains below that which could impact human health because the exposure for that measured PFAS is divided by its corresponding HBWC. For example, if the mixture only included PFNA, then under the Hazard Index approach as proposed any measured concentrations over 10.0 ng/L divided over the 10.0 ng/L HBWC would be greater than the 1.0 Hazard Index MCLG. The proposed Hazard Index MCLG was 1.0 and the HBWCs of each mixture component were as follows: 9.0 ng/L[3] for PFHxS; 10.0 ng/L for HFPO–DA; 10.0 ng/L for PFNA; and 2000.0 ng/L for PFBS (USEPA, 2023e).

b. Summary of Major Public Comments and EPA Responses

Many commenters supported the EPA's proposal to regulate a mixture of PFAS and agreed with the EPA's scientific conclusions about PFAS dose additivity and the agency's use of the Hazard Index approach to develop an MCLG for a mixture of PFHxS, PFNA, HFPO–DA, and/or PFBS. Many commenters opposed the EPA's conclusion about dose additivity and the use of the Hazard Index approach to regulate co-occurring PFAS. A few commenters opposed the EPA's use of shared or similar health endpoints/outcomes rather than a shared MOA as a basis for assessing risks of PFAS mixtures. A few commenters agreed with the EPA's decision to regulate these PFAS as a mixture (that some commenters referred to as a ''group'') and supported the EPA's conclusion about dose additivity but questioned the EPA's use of the Hazard Index and suggested alternative approaches such as development of individual MCLGs or a target organ-specific Hazard Index (TOSHI). Some commenters claimed that the EPA did not appropriately seek review from the SAB, particularly on the application of the Hazard Index as an approach to regulate PFAS under SDWA. Comments on the number of significant digits applied in the HBWCs and the Hazard Index were varied. For a discussion of comments and the EPA responses on dose additivity and similarity of toxic effects, see section III.B of this preamble. Commenters referred to the HRLs and the HBWCs interchangeably; see section III of this preamble for comments on HBWCs and the EPA's responses. Responses to the other topics raised are discussed in the following paragraphs.

The EPA disagrees with commenters that the agency did not seek adequate consultation from the EPA SAB in the development of the NPDWR. SDWA section 1412(e) requires that the EPA ''request comments'' from the SAB ''prior to proposal'' of the MCLG and NPDWR. Consistent with this statutory provision, the EPA consulted with the SAB from 2021–2022. As discussed in the proposed rule, the SAB PFAS Review Panel met virtually via a video meeting platform on December 16, 2021, and then had three (3) subsequent meetings on January 4, 6 and 7, 2022 to deliberate on the agency's charge questions, which included a question specifically focused on the utility and scientific defensibility of the Hazard Index approach in the context of mixtures risk assessment in drinking water. Another virtual meeting was held on May 3, 2022, to discuss the SAB PFAS Review Panel's draft report. Oral and written public comments were considered throughout the advisory process. The SAB provided numerous recommendations to the EPA which can be found in the SAB's final report (USEPA, 2022i). The EPA addressed the SAB's recommendations and described the EPA's responses to SAB recommendations in its *EPA Response to Final Science Advisory Board Recommendations (August 2022) on Four Draft Support Documents for the EPA's Proposed PFAS National Primary Drinking Water Regulation* (USEPA, 2023k) and also in the EPA's Response to Comments document in response to public comments on the proposed PFAS

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[3] Some commenters noted an error in the HBWC calculation for PFHxS which was reported as 9.0 ng/L in the proposal. The agency has corrected the value in this NPDWR and within the requirements under 40 CFR part 141 subpart Z. The correct HRL/HBWC for PFHxS is 10 ng/L.

NPDWR (USEPA, 2024k). Further discussion on the EPA consultations and stakeholder engagement activities can be found in section XIII of this preamble.

The agency also disagrees with commenters who contend that the EPA must seek advice from the SAB on all aspects of the NPDWR. The statute does not dictate on which scientific issues the EPA must request comment from the SAB. In this case, the EPA sought comments on four documents: *Proposed Approaches to the Derivation of a Draft Maximum Contaminant Level Goal for Perfluorooctanoic Acid (PFOA) in Drinking Water* (USEPA, 2021i); *Proposed Approaches to the Derivation of a Draft Maximum Contaminant Level Goal for Perfluorooctanesulfonic Acid (PFOS) in Drinking Water* (USEPA, 2021j); *Analysis of Cardiovascular Disease Risk Reduction as a Result of Reduced PFOA and PFOS Exposure in Drinking Water* (USEPA, 2021k); and *Draft Framework for Estimating Noncancer Health Risks Associated with Mixtures of PFAS* (USEPA, 2021e).

The approach of the EPA's *Framework for Estimating Noncancer Health Risks Associated with Mixtures of PFAS* (USEPA, 2024a) and this rule is to evaluate risks from exposure to mixtures of PFAS that elicit the same or similar adverse health effects (but with differing potencies for effect(s)) rather than similarity in MOA. This is consistent with the EPA's *Supplementary Guidance for Conducting Health Risk Assessment of Chemical Mixtures* (USEPA, 2000a) and expert opinion from the NAS National Research Council (NRC, 2008). MOA, which describes key changes in cellular or molecular events that may cause functional or structural changes that lead to adverse health effects, can be a useful metric by which risk can be assessed. It is considered a key determinant of chemical toxicity, and chemicals can often be classified by their type of toxicity pathway(s) or MOAs. However, because PFAS are an emerging chemical class, MOA data can be limited or entirely lacking for many PFAS. Therefore, the EPA's approach for assessing risks of PFAS mixtures is based on the conclusion that PFAS that share one or more adverse outcomes produce dose-additive effects from co-exposures. This evidence-based determination supports a health-protective approach that meets the statute's directive to set the MCLG at a level at which there are no known or anticipated adverse health effects and which allows for an adequate margin of safety (1412(b)(4)(A)). The EPA's evidence-based determination regarding dose additivity, based on similarity of adverse health effects rather than MOA, and use of the Hazard Index approach to assess risks of exposure to PFAS mixtures were supported by the SAB in its review of the *Draft Framework for Estimating Noncancer Health Risks Associated with Mixtures of PFAS* (USEPA, 2022i). For a detailed description of the evidence supporting dose additivity as the default approach for assessing mixtures of PFAS, see the final *Framework for Estimating Noncancer Health Risks Associated with Mixtures of PFAS* (USEPA, 2024a).

A few commenters supported the EPA's approach to assessing risks of PFAS mixtures based on similarity of toxicity effect rather than similarity in MOA. A few commenters opposed the EPA's use of same or similar adverse health effects/outcomes rather than MOA as a basis for the approach to assessing risks of PFAS mixtures and suggested that the agency is not following its own chemical mixtures guidance (USEPA, 2000a). The EPA disagrees with these commenters' assertions. The EPA's approach, to evaluate health risks of exposure to mixtures of these four PFAS based on shared or similar adverse health effects of the mixture components rather than a common MOA, is consistent with the EPA's *Supplementary Guidance for Conducting Health Risk Assessment of Chemical Mixtures* (USEPA, 2000a). Although a conclusion about dose additivity can be based on mixture components sharing a common MOA, dose additivity can also be based on ''toxicological similarity, but for specific conditions (endpoint, route, duration)'' (see the EPA's *Supplementary Guidance for Conducting Health Risk Assessment of Chemical Mixtures,* USEPA, 2000a). The EPA's *Supplementary Guidance for Conducting Health Risk Assessment of Chemical Mixtures* indicates that although basing a conclusion about dose additivity on a common MOA across mixture components is optimal, there is flexibility in the level of biological organization at which similarity among mixture components can be determined.

The EPA directly asked the SAB for feedback on this issue during its 2021 review of the EPA's draft *Framework for Estimating Noncancer Health Risks Associated with Mixtures of PFAS.* Specifically, the EPA asked the SAB, ''If common toxicity endpoint/health effect is not considered an optimal similarity domain for those PFAS with limited or no available MOA-type data, please provide specific alternative methodologies for integrating such chemicals into a component-based mixture evaluation(s)'' (USEPA, 2022i). The SAB strongly supported the EPA's approach of using a similar toxicity endpoint/health effect instead of a common MOA as a default approach for evaluating mixtures of PFAS using dose additivity and did not offer an alternative methodology. For example, the SAB panel stated that:

> The Panel agreed with use of a similar toxicity endpoint/health effect instead of a common MOA as a default approach for evaluating mixtures of PFAS. This approach makes sense because multiple physiological systems and multiple MOAs can contribute to a common health outcome. Human function is based on an integrated system of systems and not on single molecular changes as the sole drivers of any health outcome. The Panel concluded that rather than the common MOA, as presented in the EPA draft mixtures document, common physiological outcomes should be the defining position (USEPA, 2022i).

The SAB panel also stated:

> Furthermore, many PFAS, including the four used in the examples in the draft EPA mixtures document and others, elicit effects on multiple biological pathways that have common adverse outcomes in several biological systems (*e.g.,* hepatic, thyroid, lipid synthesis and metabolism, developmental and immune toxicities) (USEPA, 2022i).

Some commenters expressed support for the EPA's proposed Hazard Index approach to regulating a mixture of one or more of the four PFAS in drinking water. The commenters also stated that occurrence and co-occurrence of these four PFAS in PWSs, as well as individual and dose-additive effects of these PFAS, justify the general Hazard Index approach. The EPA agrees that the general Hazard Index approach is the most scientifically sound and health-protective approach to deriving a PFAS mixtures MCLG which considers both their dose additive health concerns and co-occurrence in drinking water (see additional discussion in the following paragraphs).

Some commenters opposed the EPA's use of a general Hazard Index as opposed to a target organ-specific Hazard Index (TOSHI) and suggested the use of a TOSHI instead. As discussed in this section, the EPA disagrees with these comments because the use of the general Hazard Index approach to develop an MCLG for a mixture of PFHxS, PFNA, HFPO–DA, and/or PFBS is scientifically sound, supported by external peer review (SAB), and consistent with the EPA's *Supplementary Guidance for Conducting Health Risk Assessment of Chemical Mixtures* (USEPA, 2000a).

The EPA considered the two main types of Hazard Index approaches: (1)

the general Hazard Index, which allows for component chemicals in the mixture to have different health effects or endpoints as the basis for their toxicity reference values (*e.g.,* RfDs, minimal risk levels), and (2) the TOSHI, which relies on toxicity reference values based on the same specific target organ or system effects (*e.g.,* effects on the liver or thyroid; effects on developmental or reproductive systems) (USEPA, 2000a). The general Hazard Index approach uses the most health-protective RfD (or minimal risk levels) available for each mixture component, irrespective of whether the RfDs for all mixture components are based on effects in the same target organs or systems. These ''overall'' RfDs (as they are sometimes called) are protective of all other adverse health effects because they are based on the most sensitive known endpoints as supported by the weight of the evidence. As a result, this approach is protective of all types of toxicity/adverse effects, and thus ensures that the MCLG is the level at and below which there are no known or anticipated adverse human health effects with an adequate margin of safety with respect to certain PFAS mixtures in drinking water. The TOSHI produces a less health protective indicator of risk than the general Hazard Index because the basis for the component chemical toxicity reference values has been limited to a specific target organ or system effect, which may occur at higher exposure levels than other effects (*i.e.,* be a less sensitive endpoint). Additionally, since a TOSHI relies on toxicity reference values aggregated for the same specific target organ or system endpoint/effect, an absence or lack of data on the specific target organ or system endpoint/effect for a mixture component may result in that component not being adequately accounted for in this approach (thus, underestimating health risk of the mixture). A TOSHI can only be derived for those PFAS for which the same target organ or system endpoint/effect-specific RfDs have been calculated. Many PFAS have data gaps in epidemiological or animal toxicological dose-response information for multiple types of health effects, thus limiting derivation of target organ-specific toxicity reference values; target organ-specific toxicity reference values are not currently available for PFHxS, PFNA, HFPO–DA, and PFBS. The EPA's *Supplementary Guidance for Conducting Health Risk Assessment of Chemical Mixtures* recognizes the potential for organ- or system-specific data gaps and supports use of overall RfDs in a general Hazard Index approach, stating, ''The target organ toxicity dose (TTD) is not a commonly evaluated measure and currently there is no official EPA activity deriving these values, as there is for the RfD and RfC'' . . . ''Because of their much wider availability than TTDs, standardized development process including peer review, and official stature, the RfD and RfC are recommended for use in the default procedure for the HI'' (USEPA, 2000a). The EPA determined that the general Hazard Index approach is the most scientifically defensible and health protective approach for considering PFAS mixtures in this rule because it is protective of all adverse health effects rather than just those associated with a specific organ or system, consistent with the statutory definition of MCLG.

The EPA directly asked the SAB about the utility and scientific defensibility of the general Hazard Index approach (in addition to other methods, including TOSHI) during the 2021 review of the EPA's draft *Framework for Estimating Noncancer Health Risks Associated with Mixtures of PFAS.* Specifically, the EPA asked the SAB to ''Please provide specific feedback on whether the HI approach is a reasonable methodology for indicating potential risk associated with mixtures of PFAS. If not, please provide an alternative;'' and ''Please provide specific feedback on whether the proposed HI methodologies in the framework are scientifically supported for PFAS mixture risk assessment'' (USEPA, 2022i). In its report (USEPA, 2022i), the SAB stated its support for the general Hazard Index approach:

> In general, the screening level Hazard Index (HI) approach, in which Reference Values (RfVs) for the mixture components are used regardless of the effect on which the RfVs are based, is appropriate for initial screening of whether exposure to a mixture of PFAS poses a potential risk that should be further evaluated. Toxicological studies to inform human health risk assessment are lacking for most members of the large class of PFAS, and mixtures of PFAS that commonly occur in environmental media, overall. For these reasons, the HI methodology is a reasonable approach for estimating the potential aggregate health hazards associated with the occurrence of chemical mixtures in environmental media. The HI is an approach based on dose additivity (DA) that has been validated and used by the EPA. The HI does not provide quantitative risk estimates (*i.e.,* probabilities) for mixtures, nor does it provide an estimate of the magnitude of a specific toxicity. This approach is mathematically straightforward and may readily identify mixtures of potential toxicological concern, as well as identify chemicals that drive the toxicity within a given mixture.

A few commenters stated that it is inappropriate to use the general Hazard Index in the context of a drinking water rule because it is a screening tool. The EPA guidance (*e.g., Risk Assessment Guidance for Superfund* [RAGS], USEPA, 1991b) and the SAB does characterize the general Hazard Index as appropriate for screening, but the SAB did not say that the methodology's use was limited to screening, nor that the agency would or should be prohibited from considering its use in any regulatory or nonregulatory application. The general Hazard Index is a well-established methodology that has been used for several decades in at least one other regulatory context to account for dose additivity in mixtures. The EPA routinely uses the Hazard Index approach to consider the risks from multiple contaminants of concern in the Remedial Investigations and Feasibility Studies for cleanup sites on the Superfund National Priorities List under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA). Noncarcinogenic effects are summed to provide a Hazard Index that is compared to an acceptable index, generally 1. This procedure assumes dose additivity in the absence of information on a specific mixture. These assessments of hazards from multiple chemical exposures are important factors to help inform the selection of remedies that are ultimately captured in the Superfund Records of Decision. Moreover, the EPA has determined that in the context of SDWA, the Hazard Index is also an appropriate methodology for determining the level at and below which there are no known or anticipated adverse human health effects with an adequate margin of safety with respect to certain PFAS mixtures in drinking water. The Hazard Index approach is the most practical approach for establishing an MCLG for PFAS mixtures that meets the statutory requirements outlined in section 1412(b)(1)(A) of SDWA. This is because the Hazard Index assesses the exposure level of each component PFAS relative to its HBWC, which is based on the most sensitive known adverse health effect (based on the weight of evidence) and considers sensitive population(s) and life stage(s) as well as potential exposure sources beyond drinking water. Furthermore, the Hazard Index accounts for dose additive health concerns by summing the hazard contribution from each mixture component to ensure that the mixture is not exceeding the level below which there are no known or anticipated adverse health effects and allows for an adequate margin of safety. In addition, given the temporal and spatial variability of PFAS occurrence in drinking water across the nation (USEPA, 2024b), this methodology allows the EPA to regulate these chemicals in drinking water by taking into account site-specific data at each PWS. Component PFAS HQs (hazard quotients) are expected to differ across time and space depending on the actual measured concentrations of each of the four PFAS at each PWS. This approach thus allows for flexibility beyond a one-size-fits-all approach and is tailored to address risk at each PWS. The EPA has made a final regulatory determination for mixtures of two or more of these PFAS. The EPA's application of the Hazard Index approach to regulate such mixtures accounts for the dose additivity that was the basis for the EPA's final determination to regulate such mixtures.

A Hazard Index greater than 1 is generally regarded as an indicator of adverse health risks associated with a specific level of exposure to the mixture; a Hazard Index less than or equal to 1 is generally regarded as not being associated with any appreciable risk (USEPA, 1986; USEPA,1991b; USEPA, 2000a). Thus, in the case of this drinking water rule, a Hazard Index greater than 1 indicates that occurrence of two or more of these four component PFAS in a mixture in drinking water exceeds the health protective level(s) (*i.e.,* HBWC(s)), indicating health risks.

The EPA proposed a Hazard Index MCLG of 1.0, expressed with two significant digits. The EPA's proposal expressed the HBWCs to the tenths place, as follows: 9.0 ng/L for PFHxS, 10.0 ng/L for HFPO–DA; 10.0 ng/L for PFNA; and 2000.0 ng/L for PFBS. The EPA's draft Hazard Index MCLG document expressed all of the HBWCs with one significant digit (9, 10, 10, 2000 ng/L, respectively) (USEPA, 2023e). A few commenters supported the use of two significant digits for the HBWCs, individual HQs, and the Hazard Index MCLG and stated that the use of two significant digits would not be expected to result in issues related to analytical methods precision. One commenter supported using all digits of precision in calculations but rounding to two significant digits for the final reported value of the Hazard Index, noting that the number of significant digits used only affects rounding during steps prior to the point at which a Hazard Index MCL is reached. Commenters noted the importance of clearly communicating the number of significant digits to be used in the documents, and that the choice of the number of significant digits could impact implementation of an MCL based on the Hazard Index. For example, a Hazard Index of 1 (*i.e.,* using one significant digit) would not be exceeded unless the value is calculated to be at 1.5 or above. Alternatively, a Hazard Index of 1.0 (reporting with more than one significant digit) would be exceeded when the Hazard Index is calculated to be 1.05 or above. For additional discussion on significant digit usage, please see sections V and VIII.

A few commenters did not support more than a single significant digit for the HBWCs and Hazard Index MCLG, with some stating that using two or more significant digits for the Hazard Index contradicts the EPA chemical mixtures guidance (USEPA, 2000a) and the RAGS (USEPA, 1991b). The EPA agrees that one (1) significant digit is appropriate for the HBWCs and the Hazard Index MCLG (*i.e.,* 1 rather than 1.0, as in the proposal) because although there is sufficient analytical precision for two significant digits at these concentrations, the RfVs (RfDs and minimal risk levels) used to derive the HBWCs have one significant digit. According to the EPA chemical mixtures guidance (USEPA, 2000a), ''Because the RfDs (and by inference the TTDs) are described as having precision no better than an order of magnitude, the HI should be rounded to no more than one significant digit.'' This approach of using a Hazard Index of 1 is consistent with agency chemical mixtures guidance (USEPA, 1986; USEPA, 2000a) and RAGS (USEPA, 1991b; USEPA, 2018c). The EPA's *Risk Assessment Guidance for Superfund Volume 1 Human Health Evaluation Manual* states, ''For noncarcinogenic effects, a concentration is calculated that corresponds to an HI of 1, which is the level of exposure to a chemical from all significant exposure pathways in a given medium below which it is unlikely for even sensitive populations to experience adverse health effects,'' and ''The total risk for noncarcinogenic effects is set at an HI of 1 for each chemical in a particular medium'' (USEPA, 1991b). Finally, ''Cancer risk values and hazard index (HI) values may express more than one significant figure, but for decision-making purposes one significant figure should be used'' (USEPA, 2018c).

c. Final Rule

The EPA has made a final determination to regulate mixtures containing two or more of PFHxS, PFNA, HFPO–DA, and/or PFBS. For the final determination, the EPA's evaluation utilized an HRL as part of a general Hazard Index approach (for additional discussion on the EPA's *Final Regulatory Determinations,* please see section III of this preamble). The EPA's proposal included individual preliminary regulatory determinations for PFHxS, PFNA, HFPO–DA, and PFBS and a mixture regulatory determination for mixtures of those PFAS. The EPA's proposal addressed these regulatory determinations through the Hazard Index MCLG and MCL that would apply to a mixture containing one or more of PFHxS, PFNA, HFPO–DA, and PFBS. If two or more of these PFAS were present then the MCLG and MCL would account for dose additivity of all of the contaminants present, but if only one of the contaminants were present then the Hazard Index would operate as an individual MCLG and MCL. In this final rule, the EPA is promulgating individual MCLGs and MCLs to address the individual final regulatory determinations (PFHxS, PFNA, and HFPO–DA) and is promulgating a Hazard Index MCLG and MCL to address the final mixtures regulatory determination for two or more Hazard Index PFAS (PFHxS, PFNA, HFPO–DA, and PFBS) present.

The EPA used the same general Hazard Index approach for the mixture MCLG. In the general Hazard Index approach, individual PFAS HQs are calculated by dividing the measured concentration of each component PFAS in water (*e.g.,* expressed as ng/L) by the corresponding HBWC for each component PFAS (*e.g.,* expressed as ng/L), as shown in the following equation (and described in USEPA, 2024f). For purposes of this NPDWR, the EPA is using the term ''health-based water concentration'' or ''HBWC'' given its role in calculating the Hazard Index (see the Executive Summary of this preamble). The EPA notes that the Hazard Index MCLG applies to the entire mixture but the EPA's technical justification for the HBWCs for the mixture components is the same as for the individual MCLGs provided in this rule. In this final rule, component PFAS HQs are summed across the PFAS mixture to yield the Hazard Index MCLG. The final PFAS mixture Hazard Index MCLG is set at 1 (one significant digit). A Hazard Index greater than 1 (rounded to one significant digit) indicates that exposure (*i.e.,* PFAS occurrence in drinking water) exceeds the health protective level (*i.e.,* HBWC) for two or more of the individual PFAS mixture components, and thus indicates health risks. The Hazard Index MCLG ensures that even when the individual components are below a level of concern, the components when added together in the mixture do not result in a mixture that itself exceeds a level of concern. A Hazard Index less than or equal to 1 indicates that occurrence of these four PFAS in drinking water does not exceed the health protective level and is therefore generally regarded as unlikely to result in any appreciable risk (USEPA, 1986; USEPA, 1991b; USEPA, 2000a). For more details, please see USEPA (2024a; USEPA, 2024f). The final Hazard Index MCLG for a mixture of PFHxS, PFNA, HFPO–DA, and/or PFBS is derived as follows:

$$HI\ MCLG\ =\left(\frac{[HFPO-DA_{water}]}{[HFPO-DA_{HBWC}]}\right)+\left(\frac{[PFBS_{water}]}{[PFBS_{HBWC}]}\right)+\left(\frac{[PFNA_{water}]}{[PFNA_{HBWC}]}\right)+\left(\frac{[PFHxS_{water}]}{[PFHxS_{HBWC}]}\right)=1$$

$$HI\ MCLG\ =\left(\frac{[HFPO-DA_{ng/L}]}{[10\,ng/L]}\right)+\left(\frac{[PFBS_{ng/L}]}{[2000\,ng/L]}\right)+\left(\frac{[PFNA_{ng/L}]}{[10\,ng/L]}\right)+\left(\frac{[PFHxS_{ng/L}]}{[10\,ng/L]}\right)=1$$

Where

$[PFAS_{water}]$ = the measured component PFAS concentration in water and

$[PFAS_{HBWC}]$ = the HBWC of a component PFAS.

2. MCLG Derivation for PFHxS, PFNA, and HFPO–DA

a. Proposal

As described in section IV.B.1.a of this preamble, in March 2023, the EPA proposed a Hazard Index MCLG to protect public health from exposure to mixtures of PFHxS, PFNA, HFPO–DA, and PFBS, four PFAS that affect many similar health endpoints/outcomes and that occur and co-occur in drinking water. At that time, the EPA also considered setting individual MCLGs for these PFAS either instead of or in addition to using a mixtures-based approach for PFHxS, PFNA, HFPO–DA, and PFBS. The EPA ultimately proposed the Hazard Index approach for establishing an MCLG for a mixture of these four PFAS.

b. Summary of Major Public Comments and EPA Responses

Several commenters favored finalization of individual MCLGs (and MCLs) for some or all of the PFAS included in the proposed Hazard Index, with or without a Hazard Index approach to address mixtures of these PFAS. Specifically, commenters supported establishing individual MCLGs for PFHxS, PFNA, HFPO–DA, and PFBS because they questioned the EPA's scientific conclusions regarding PFAS dose additivity and raised concerns about potential risk communication issues and confusion about the EPA's use of the Hazard Index to establish drinking water standards (for additional discussion on MCLs, please see section V of this preamble). The EPA agrees with commenters who favored finalization of individual MCLGs for some of the PFAS included in the Hazard Index, and to do so in addition to the Hazard Index MCLG being finalized for the mixture of the four PFAS. The EPA believes this provides clarity for purposes of implementation of the rule. The EPA is finalizing individual MCLGs for PFHxS, PFNA, and HFPO–DA (for additional discussion on the final regulatory determinations, please see section III of this preamble). Regarding risk communication and potential confusion about the use of the Hazard Index, the EPA acknowledges that effective risk communication is important, and the agency will develop communication materials to facilitate understanding of all aspects of this NPDWR, including the Hazard Index MCL (for additional discussion on MCLs, please see section V of this preamble). The EPA has provided language for consumer notifications as part of CCR (see section IX of this preamble).

One commenter stated that developing individual MCLGs (and MCLs) in addition to the Hazard Index mixture MCLG (and MCL) would have no practical impact, since an exceedance of an HBWC for an individual PFAS within a mixture would result in an exceedance of the Hazard Index even if none of the other PFAS included in the Hazard Index are detected. The EPA clarifies the final rule promulgates individual MCLs for PFHxS, PFNA and HFPO–DA as well as a mixture Hazard Index MCL for two or more of these PFAS and PFBS. There may be a practical impact of these individual MCLs (for PFHxS, PFNA and HFPO–DA) where one of these three PFAS occur in isolation (*i.e.,* without one of the other four Hazard Index PFAS present) above their individual MCLs. The EPA notes that this regulatory structure is consistent with the intended effect of the proposed regulation, where as proposed, a single PFAS above its HBWC would have caused an exceedance of the MCL. Based on public comment, the EPA has restructured the rule such that two or more of these regulated PFAS would be necessary to cause an exceedance of the Hazard Index and instead will regulate individual exceedances of PFNA, PFHxS, and HFPO–DA as individual MCLs to improve risk communication. Risk communication is an important focus for water systems and the EPA believes that finalizing individual MCLs for PFHxS, PFNA, and HFPO–DA can support risk communication as utilities and the public may be more familiar with this regulatory framework. Additionally, the final individual MCLs for PFHxS, PFNA and HFPO–DA will address and communicate health concerns for these compounds where they occur in isolation. At the same time, since those individual MCLs do not address additional risks from co-occurring PFAS, the EPA is finalizing a Hazard Index MCL that provides a framework to address and communicate dose additive health concerns associated with mixtures of PFHxS, PFNA, HFPO–DA, and PFBS that co-occur in drinking water. For the EPA's discussion on the practical impact of the establishment of stand-alone standards in lieu of or in addition to the Hazard Index MCL, please see sections V and IX.A of this preamble. The EPA's discussion on the practical impact of the establishment of stand-alone standards in lieu of or in addition to the Hazard Index MCL, please see sections V and IX.A of this preamble.

A few commenters questioned why the EPA is developing an NPDWR for contaminants that do not have EPA Drinking Water Health Advisories (PFHxS, PFNA), and stated that the EPA should wait to propose an NPDWR for PFHxS and PFNA until after Health Advisories are finalized for these PFAS. The EPA disagrees with this comment. Health Advisories are not a pre-requisite for an NPDWR under SDWA and there is nothing in the statute or the EPA's historical regulatory practice that suggests that the agency must or should delay regulation of a contaminant in order to develop a health advisory first.

c. Final Rule

As described in section III of this preamble, the EPA has made a final determination to individually regulate PFHxS, PFNA, and HFPO–DA.

The EPA is finalizing individual MCLGs for PFHxS, PFNA, and HFPO–DA as follows: PFHxS MCLG = 10 ng/L; HFPO–DA MCLG = 10 ng/L; and PFNA MCLG = 10 ng/L. The technical basis for why each of these levels satisfies the statutory definition for MCLG is described in section III of this preamble (and is the same technical basis the EPA used to explain the levels identified as the HBWCs). These MCLGs are expressed with one significant digit and are based on an analysis of each chemical's toxicity (*i.e.,* RfD/minimal risk level) and appropriate exposure factors (*i.e.,* DWI–BW, RSC) (USEPA, 2024f).

The EPA is deferring its individual regulatory determination for PFBS and not finalizing an individual MCLG for PFBS at this time (please see section III of this preamble, *Final Regulatory Determinations for Additional PFAS,* for further information).

V. Maximum Contaminant Levels

Under current law and as described in the proposed rule (USEPA, 2023f), the Environmental Protection Agency (EPA) establishes drinking water standards through a multi-step process. *See* S. Rep. No. 169, 104th Cong., 1st Sess. (1995) at 3. First, the agency establishes a non-enforceable Maximum Contaminant Level Goals (MCLG) for the contaminant in drinking water at a level which no known or anticipated adverse effects to the health of persons will occur and which allow for an adequate margin of safety. Second, the agency generally sets an enforceable Maximum Contaminant Level (MCL) as close to that public health goal as feasible, taking costs into consideration.

In this second step, consistent with the definition of ''feasible'' in section 1412(b)(4)(D), the EPA evaluates the availability and performance of Best Available Technologies (BATs) for treating water to minimize the presence of the contaminant consistent with the MCLG (see section X for additional discussion on BATs) as well as the costs of applying those BATs to large metropolitan water systems when treating to that level (1412(b)(4)(E) and (5)).[4] The definition of ''feasible'' means *feasible* with the use of the best technology . . . ''which includes consideration of the analytical limits of best available treatment and testing technology.'' *see* S. Rep. No. 169, 104th Cong., 1st Sess. (1995) at 3; *see also* section 1401(1)(C)(i) stating that a NPDWR includes an MCL only ''if, in the judgment of the Administrator, it is economically and technologically feasible to ascertain the level of such contaminant in water in public water systems.'' In addition, the MCL represents ''the maximum permissible level of a contaminant in water which is delivered to any user of a public water system,'' section 1401(3). Thus, in setting the MCL level, the EPA also identifies the level at which it is technologically feasible to measure the contaminant in the public water system. To identify this level, the EPA considers (1) the availability of analytical methods to reliably quantify levels of the contaminants in drinking water and (2) the lowest levels at which contaminants can be reliably quantified within specific limits of precision and accuracy during routine laboratory operating conditions using the approved methods (known as the practical quantitation levels (PQLs)). The ability of laboratories to measure the level of the contaminant with sufficient precision and accuracy using approved methods is essential to ensure that any public water system nationwide can monitor, determine compliance, and deliver water that does not exceed the maximum permissible level of a contaminant in water to any of its consumers. (See section VII of this preamble for additional discussion on analytical methods and PQLs for the per- and polyfluoroalkyl substances (PFAS) regulated in this rule.)

In practice this means that where the MCLG is zero, the EPA typically sets MCLs at the PQLs when treatment is otherwise feasible, based on cost and treatment availability, because the PQL is the limiting factor. Conversely, for contaminants where the MCLG is higher than the PQL, the EPA generally sets the MCL at the MCLG when treatment is otherwise feasible, based on costs and treatment availability, because the PQL is not a limiting factor.

The Safe Drinking Water Act (SDWA) defines an MCL as ''the maximum permissible level of a contaminant in water which is delivered to any user of a public water system.'' Like the MCLG, SDWA does not dictate that the MCL take a particular form; however, given this definition, an MCL establishes a ''maximum permissible level of a contaminant in water'' and as a practical matter the identified ''level'' must be capable of being validated so that it can be determined whether that public water systems are delivering water to any user meeting or exceeding that ''level.''

A. PFOA and PFOS

1. Proposal

In the March 2023 proposal, the EPA proposed individually enforceable MCLs for PFOA and PFOS at the PQL which is 4.0 ng/L (USEPA, 2023f). Section 1412(b)(4)(E) of SDWA requires that the agency ''list the technology, treatment techniques, and other means which the Administrator finds to be feasible for purposes of meeting [the MCL],'' which are referred to as Best Available Technologies (BATs). The EPA found multiple treatment technologies to be effective and available to treat PFOA and PFOS to at or below the proposed standards (please see and section X (10) of this preamble and USEPA, 2024l for additional discussion on feasible treatment technologies including BAT/SSCT identification and evaluation). In addition, the EPA found that there are analytical methods available to reliably quantify PFOA and PFOS at the PQL. The EPA requested comment on regulatory alternatives for both compounds at 5.0 ng/L and 10.0 ng/L. The EPA also requested comment on whether setting the MCL at the PQL for PFOA and PFOS is implementable and feasible.

2. Summary of Major Public Comments and EPA Responses

The EPA received many comments that strongly support the proposed MCLs of 4.0 ng/L and the agency's determination that the standards are as close as feasible to the MCLG. These commenters request the agency to finalize the standards as expeditiously as possible. Consistent with these comments, through this action, the agency is establishing drinking water standards for PFOA and PFOS (and four other PFAS) to provide health protection against these contaminants found in drinking water.

Many commenters assert that implementation of the PFOA and PFOS standards would be challenging because the MCLs are set at the PQLs for each compound, and some commenters recommended alternative standards (*e.g.,* 5.0 ng/L or 10.0 ng/L). These commenters contend that by setting the

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[4] Based on legislative history, the EPA interprets ''taking cost into consideration'' in section 1412(b)(4)(D) to be limited to ''what may be reasonably be afforded by large metropolitan or regional public water systems.'' H.R. Rep. No 93–1185 (1974), *reprinted in* 1974 U.S.C.C.A.N. 6454, 6470–71.

MCLs at the PQLs, utilities would not be able to reliably measure when the concentration of contaminants in their drinking water is approaching the MCLs. Some of these commenters suggest that having a buffer between the PQLs and the MCLs may allow utilities to manage treatment technology performance more efficiently because utilities generally aim to achieve lower than the MCLs to avoid a violation and that this buffer would provide some level of operational certainty for systems treating for PFAS. The EPA disagrees that the PFOA and PFOS standards are not implementable because the MCLs are set at their respective PQLs.

As the agency noted in the proposed rule preamble, the EPA has promulgated, and both the EPA and water systems have successfully implemented, several NPDWRs with MCLs equal to the contaminant PQLs. As examples, in 1987, the EPA finalized the Phase I Volatile Organic Compounds (VOC) rule (USEPA, 1987), where the agency set the MCL at the PQL for benzene, carbon tetrachloride, trichloroethylene, vinyl chloride, and 1,2-dichloroethane (52 FR 25690). Other examples where MCLs were set at the PQL include benzo(a)pyrene, di(2-ethylhexyl) phthalate, dioxin, dichloromethane, hexachlorobenzene, and PCBs (see USEPA, 1991c and USEPA, 1992). Some commenters at the time stated they believed implementation would be challenging because the MCLs were set at the PQL in these examples; however, the EPA notes that those rules have been implemented successfully despite commenters initial concerns. The agency does not agree with commenters that operational flexibility (*i.e.,* the inclusion of a 'buffer' between the PQL and MCL) is relevant for purposes of setting an MCL. That is because the PQL is the lowest level that can be reliably achieved within specified limits of precision and accuracy and is therefore the metric by which the agency uses to evaluate the most feasible MCL pursuant to SDWA requirements. Considerations for operational flexibility may be relevant to other parts of the rule, such as determining monitoring and compliance with the rule. First, for purposes of determining compliance with the MCL, water systems must calculate the running annual average (RAA) of results, which could allow some results to exceed 4.0 ng/L for single measurements if the overall annual average is below the MCL. In other words, there is a buffer built into determining compliance with the MCL. Second, when calculating the RAA, zero will be used for results less than the PQL which provides an additional analytic buffer for utilities in their compliance calculations. This monitoring and compliance framework allows for temporal fluctuations in concentrations that may occur because of unexpected events such as premature PFOA and PFOS breakthrough or temporary elevated source water concentrations. Thus, periodic occurrences of PFOA or PFOS that are slightly above the PQLs do not necessarily result in a violation of the MCL if other quarterly samples are below the PQL. The agency notes that in general, PQLs are set above the limit of detection; for PFAS specifically, all the PQLs are well above their limits of detection. The PQL is also different than detection limits because the PQL is set considering a level of precision, accuracy, and quantitation. Systems may be able to use sample results below the PQL to understand whether PFOA and PFOS are present. While the EPA has determined that results below the PQL are insufficiently precise for determining compliance with the MCL, results below the PQL can be used to determine analyte presence or absence in managing a system's treatment operations and to determine monitoring frequency. See discussion in section VII of this preamble for further discussion of the PQL, results below the PQL, and how those results provide useful information.

Some commenters contend that the PQLs for PFOA and PFOS are not set at an appropriate level (*e.g.,* the PQLs are either too high or too low for laboratories to meet). Specifically, these commenters question whether enough laboratories have the ability to analyze samples at 4.0 ng/L and, as a result, contend it is not a ''reasonable quantitation level.'' The EPA disagrees with commenters who suggest the PQLs for PFOA and PFOS are not set at an appropriate level or that they should be either higher or lower levels than that proposed. As discussed above and in the March 2023 proposal, the EPA derives PQLs that reflect the level of contaminants that laboratories can reliably quantify within specific limits of precision and accuracy during routine laboratory operating conditions. The ability to reliably measure is an important consideration for feasibility to ensure that water systems nationwide can monitor and dependably comply with the MCLs and deliver drinking water that does not exceed the maximum permissible level. In the rule proposal (USEPA, 2023f), the EPA explained that the minimum reporting levels under UCMR 5 reflect ''a minimum quantitation level that, with 95 percent confidence, can be achieved by capable lab analysts at 75 percent or more of the laboratories using a specified analytical method'' (USEPA, 2022k). The PQLs for the regulated PFAS are based on the UCMR 5 minimum reporting levels. The EPA calculated the UCMR 5 minimum reporting levels using quantitation-limit data from multiple laboratories participating in multi-lab method validation studies conducted in the 2017–2019 timeframe, prior to the UCMR 5 Laboratory Approval Program (see appendix B of USEPA, 2020b). The calculations account for differences in the capability of laboratories across the country. Laboratories approved to analyze UCMR samples must demonstrate that they can consistently make precise measurements of PFOA and PFOS at or below the established minimum reporting levels. Therefore, the EPA finds that the UCMR 5 minimum reporting levels are appropriate for using as PQLs for this rule: the EPA estimates that laboratories across the nation can precisely and accurately measure PFOA and PFOS at this quantitation level. After reviewing data from laboratories that participated in the minimum reporting level setting study under UCMR 5 and in consideration of public comment, the EPA finds that the minimum reporting levels set in UCMR 5 of 4.0 ng/L for PFOA and PFOS, that are also the PQLs, are as close as feasible to the MCLG. While lower quantitation levels may be achievable for some laboratories, it has not been demonstrated that these lower quantitation levels can be achieved for ''at 75 percent or more of the laboratories using a specified analytical method'' across laboratories nationwide. Moreover, though the EPA is confident of sufficient laboratory capacity to implement this PFAS National Primary Drinking Water Regulation (NPDWR) as finalized, a lower PQL could potentially limit the number of laboratories available to support analytical monitoring that would be otherwise available to support analytical monitoring with PFOA and PFOS PQLs of 4.0 ng/L.

In the proposal, the EPA discussed how utilities may be able to use sample results below the PQL to determine analyte presence or absence in managing their treatment operations; however, a few commenters contend that this is not practical to determine compliance with the MCL as these values are less precise and violations may result in expensive capital improvements. Commenters are conflating two different issues. While commenters are referring to quantitation of a sampling result for compliance with the rule, the EPA's discussion on results below the PQL refers to determining simple presence or absence of a contaminant for other purposes. Sampling results below the PQL may not have the same precision as a sampling result at or above the PQL but they are useful for operational purposes such as understanding that PFOA and PFOS may be present, which can inform treatment decisions and monitoring frequency. For example, a utility may use sampling results below 4.0 ng/L as a warning that they are nearing the PFOA and PFOS MCLs of 4.0 ng/L prior to an exceedance. Then, the utility can make informed treatment decisions about managing their system (*e.g.,* replacing GAC). Additionally, the EPA evaluated data submitted as part of the UCMR 5 Laboratory Approval Program (LAP) and found that 47 of 53 laboratories (89 percent) that applied for UCMR 5 approval generated a minimum reporting level confirmation at 2 ng/L (one-half the proposed MCL) or less for Method 533 (USEPA, 2022j). This suggests that the majority of laboratories with the necessary instrumentation to support PFAS monitoring have the capability to provide useful screening measurement results below the PQL. Further, as discussed in section VII of this preamble, all labs are required per the approved methods to demonstrate whether laboratory reagent blank (LRB) quality control (QC) samples have background concentrations of less than one-third the minimum reporting level (*i.e.,* the minimum concentration that can be reported as a quantitated value for a method analyte in a sample following analysis). Therefore, for a laboratory to be compliant with the methods, they must be able to detect, not necessarily quantify, analytes at or above ⅓ the minimum reporting level.

The EPA agrees with commenters that it is inappropriate to make potentially costly compliance decisions based on measurements below the PQL because they do not have the same level of precision and accuracy as results at or above the PQL. As previously discussed, for MCL compliance purposes, results less than the PQL will be recorded as zero. For additional details on monitoring and compliance requirements, please see section VIII of this preamble.

Some commenters argue that the EPA did not sufficiently consider cost in the agency's feasibility analysis of the proposed MCLs and therefore disagreed with the EPA that the standards are feasible. In particular, these commenters suggest that the agency did not adequately consider costs associated with implementation (*e.g.,* costs for labor, materials, and construction of capital improvements) and compliance (*e.g.,* costs to monitor) with the proposed MCLs. Based on these factors, many of these commenters suggest either raising the MCLs or re-proposing the standard in its entirety. The EPA did consider these costs and therefore disagrees with commenters' assertions that the agency did not consider these issues in establishing the proposed MCLs for PFOA and PFOS (USEPA, 2024g; USEPA, 2024l; USEPA, 2024m). The EPA considers whether these costs are reasonable based on large metropolitan drinking water systems. H.R. Rep. No 93–1185 (1978), *reprinted in* 1974 U.S.C.C.A.N. 6454, 6470–71. The EPA considered costs of treatment technologies that have been demonstrated under field conditions to be effective at removing PFOA and PFOS and determined that the costs of complying with an MCL at the PQL of 4.0 are reasonable for large metropolitan water systems at a system and national level (USEPA, 2024e; USEPA, 2024g). To designate technologies as BATs, the EPA evaluated each technology against six BAT criteria, including whether there is a reasonable cost basis for large and medium water systems. The EPA evaluated whether the technologies are currently being used by systems, whether there were treatment studies available with sufficient information on design assumptions to allow cost modeling, and whether additional research was needed (USEPA, 2024l). In considering the results of this information, the EPA determined that these costs are reasonable to large metropolitan water systems.

Pursuant to SDWA section 1412(b)(4)(E)(ii), the agency also evaluated ''technolog[ies], treatment technique[s], or other means that is affordable'' for small public water systems. In this evaluation, the agency determined that the costs of small system compliance technologies (SSCTs) to reach 4.0 ng/L are affordable for households served by small drinking water systems. Additionally, the EPA notes that SDWA section 1412(b)(4)(D) states that ''granular activated carbon is feasible for the control of synthetic organic chemicals'' which the agency lists as a BAT for this rule (section X). All PFAS, including PFOA and PFOS, are SOCs, and therefore, GAC is BAT as defined by the statute. For additional discussion on BATs and SSCTs, please see section X of this preamble.

Some commenters disagreed with the EPA's determination that the rule is feasible under SDWA asserting that there is insufficient laboratory capacity and other analytic challenges to measure samples at these thresholds. As described above in the agency's approach toward evaluating feasibility, the EPA assesses (1) the availability of analytical methods to reliably quantify levels of the contaminants in drinking water and (2) the lowest levels at which contaminants can be reliably quantified within specific limits of precision and accuracy during routine laboratory operating conditions using the approved methods (*i.e.,* the PQLs). This framework inherently considers both the capacity and capability of labs available to meet the requirements of the NPDWR. Based on the EPA's analysis of these factors, the EPA disagrees with commenter assertions that there is insufficient laboratory capacity at this time to support implementation of the NPDWR. Currently, there are 53 laboratories for PFAS methods (Method 533 or 537.1) in the EPA's Unregulated Contaminant Monitoring Rule (UCMR) 5 Laboratory Approval Program, more than double the participation in UCMR 3 (21 laboratories), with several laboratory requests to participate after the lab approval closing date. At a minimum, these 53 labs alone have already demonstrated sufficient capacity for current UCMR 5 monitoring, which requires monitoring for all systems serving above 3,300 or more persons and 800 systems serving less than 3,300 persons over a three-year period. The 21 laboratories participating in UCMR 3 provided more than sufficient capacity for that monitoring effort, which required monitoring for all systems serving greater than 10,000 persons and 800 systems serving less than 10,000. Further, a recent review of state certification and third-party accreditation of laboratories for PFAS methods found an additional 25 laboratories outside the UCMR 5 LAP with a certification or accreditation for EPA Method 533 or 537.1. Additionally, as has happened with previous drinking water regulations, the EPA anticipates laboratory capacity to grow once the rule is finalized to include an even larger laboratory community, as the opportunity for increased revenue by laboratories would be realized by filling the analytical needs of the utilities (USEPA, 1987; USEPA, 1991c; USEPA, 1991d; USEPA, 1992; USEPA, 2001). Finally, with the use of a reduced monitoring schedule to once every three years for eligible systems, and the ability for systems that are reliably and consistently below the MCLs of 4.0 ng/L to only monitor once per year, the EPA anticipates that the vast majority of utilities may be able to take advantage of reduced or annual monitoring, and will not require a more frequent monitoring schedule, thus easing the burden of laboratory capacity as well.

The EPA also disagrees with commenter assertions that there is insufficient laboratory capability at this time. As discussed above and in the proposed rule preamble, the EPA proposed a PQL of 4.0 ng/L for both PFOA and PFOS based on current analytical capability and from the minimum reporting levels generated for the UCMR 5 program. The EPA evaluated data submitted as part of the UCMR 5 LAP and found that 47 of 53 laboratories (89 percent) that applied for UCMR 5 approval generated a minimum reporting level confirmation at 2 ng/L (one-half the proposed MCL) or less for Method 533. The MCLs for PFOA and PFOS were also set at 4.0 ng/L as a result of the analytical capability assessment under the minimum reporting level setting study for UCMR 5, as well as consideration of other factors (*e.g.,* treatment, costs) as required under SDWA. For UCMR 5, all UCMR-approved laboratories were able to meet or exceed the PFOS and PFOA UCMR minimum reporting levels, set at 4 ng/L, the proposed MCL for both. The UCMR 5 minimum reporting levels of 4 ng/L for PFOS and PFOA are based on a multi-laboratory minimum reporting level calculation using lowest concentration minimum reporting level (LCMRL) data. The LCMRL and minimum reporting level have a level of confidence associated with analytical results. More specifically, the LCMRL calculation is a statistical procedure for determining the lowest true concentration for which future analyte recovery is predicted with 99% confidence to fall between 50 and 150% recovery (Martin et al., 2007). The multi-laboratory minimum reporting level is a statistical calculation based on the incorporation of LCMRL data collected from multiple laboratories into a 95% one-sided confidence interval on the 75th percentile of the predicted distribution referred to as the 95–75 upper tolerance limit. This means that 75% of participating laboratories will be able to set a minimum reporting level with a 95% confidence interval. The quantitation level of 4 ng/L has been demonstrated to be achieved with precision and accuracy across laboratories nationwide, which is important to ensure that systems can dependably comply with the MCL and deliver drinking water that does not exceed the maximum permissible level. The agency anticipates that these quantitation levels for labs will continue to improve over time, as technology advances and as laboratories gain experience with the PFAS Methods. The EPA's expectation is supported by the record borne out by the significant improvements in analytical capabilities for measuring certain PFAS, including PFOA and PFOS, between UCMR 3 and UCMR 5. For example, the minimum reporting levels calculated for UCMR 3 (2012–2016) were 40 ng/L and 20 ng/L for PFOS and PFOA, respectively, the minimum reporting levels calculated for UCMR 5 (2022–2025) were 4 ng/L each for PFOA and PFOS.

Some commenters recommend a different regulatory framework than what the EPA proposed to alleviate perceived implementation concerns (*e.g.,* reduce the potential of inundating laboratories or providing more time to plan and identify opportunities for source water reduction). For example, a few commenters suggest a phased-in MCL, where systems demonstrating higher concentrations are addressed first in the NPDWR, or MCL approaches where interim targets are set for compliance. Upon consideration of information submitted by commenters, particularly issues related to supply chain complications that are directly or indirectly related to the COVID–19 pandemic residual challenges, the EPA has determined that a significant number of systems subject to the rule will require an additional 2 years to complete the capital improvements necessary to comply with the MCLs for PFAS regulated under this action. Thus, the EPA also disagrees with recommendations to create a phased schedule for rule implementation based on the concentrations of PFAS detected because the EPA has granted a two-year extension for MCL compliance to all systems. For additional discussion on this extension and the EPA responses to public comment on this issue, please see section XI.D.

Some commenters argue for a lower PFOA and PFOS MCL due to the underlying health effects of these contaminants. These commenters suggest the EPA establish MCLs lower than the agency's proposed standard of 4.0 ng/L due to the capability of some laboratories to quantitate lower concentrations. Some of these commenters also argue that since PFOA and PFOS are likely human carcinogens, the EPA should consider an MCL at zero. While the EPA agrees with the health concerns posed by PFAS that are the basis for the proposed health based MCLGs for these contaminants, the agency disagrees with commenters on these alternative MCL thresholds given the EPA's consideration of feasibility as required by SDWA. These commenters did not provide evidence demonstrating the feasibility of achieving lower MCL thresholds (including an MCL at zero) consistent with SDWA requirements in establishing an MCL. For example, commenters did not provide evidence to support a lower PQL that can be consistently achieved by laboratories across the country. They also did not provide arguments supporting why the EPA should accept less than 75% of participating laboratories will be able to set a minimum reporting level with a 95% confidence interval. Thus, the agency is finalizing the MCLs for PFOA and PFOS at 4.0 ng/L (at the PQL) as this is the closest level to the MCLG that is feasible due to the ability of labs using approved analytical methods to determine with sufficient precision and accuracy whether such a level is actually being achieved. The record supports the EPA's determination that the lowest feasible MCL for PFOA and PFOS at this time is 4.0 ng/L.

A few commenters suggest the EPA did not appropriately consider disposal concerns for spent treatment media as part of the agency's feasibility determination. These commenters state that they believe disposal options are currently limited for liquid brine, reject waters resulting from RO, or solid waste from GAC treatment and that disposal capacity will be further limited should the EPA designate PFAS waste as hazardous. These commenters contend that these limitations increase operating expenses for utilities and should be factored in the establishment of the PFOA and PFOS MCLs. The EPA disagrees with these commenters that the agency did not adequately consider disposal of spent treatment media in the rule. First, disposal options for PFAS are currently available. These destruction and disposal options include landfills, thermal treatment, and underground injection. Systems are currently disposing of spent media, such as activated carbon, through thermal treatment, to include reactivation, and at landfills. While precautions should be taken to minimize PFAS release to the environment from spent media, guidance exists that explains the many disposal options with relevant precautions. See section X for further discussion. Furthermore, the EPA has provided guidance for pretreatment and wastewater disposal to manage PFAS that enters the sanitary sewer system and must be managed by publicly owned treatment works (POTWs) (USEPA, 2022d; USEPA, 2022e). As discussed in the proposed rule (USEPA, 2023f), the EPA assessed the availability of studies of full-scale treatment of residuals that fully characterize residual waste streams and disposal options. Although the EPA anticipates that designating chemicals as hazardous substances under CERCLA generally should not result in limits on the disposal of PFAS drinking water treatment residuals, the EPA has estimated the treatment costs for systems both with the use of hazardous waste disposal and non-hazardous disposal options to assess the effects of potentially increased disposal costs. Specifically, the EPA assessed the potential impact on public water system (PWS) treatment costs associated with hazardous residual management requirements in a sensitivity analysis. The EPA's sensitivity analysis demonstrates that potential hazardous waste disposal requirements may increase PWS treatment costs marginally; however, the increase in PWS costs is not significant enough to change the agency's feasibility determination nor the determination made at proposal that benefits of the rulemaking justify the costs. These estimates are discussed in greater detail in the HRRCA section of this final rule and in appendix N of the *Economic Analysis* (EA) (USEPA, 2024e). For the discussion on management of treatment residuals and additional responses to stakeholder concerns on this topic, please see section X of this preamble. While beyond the scope of this rule, the EPA further notes that the agency is proposing to amend its regulations under the Resource Conservation and Recovery Act (RCRA) by adding nine specific per-and polyfluoroalkyl substances (PFAS), their salts, and their structural isomers, to the list of hazardous constituents at 40 CFR part 261, appendix VIII (89 FR 8606). The scope of the proposal is limited and does not contain any requirements that would impact disposal of spent drinking water treatment residuals. This is because listing these PFAS as RCRA hazardous constituents does not make them, or the wastes containing them, RCRA hazardous wastes. The principal impact of the proposed rule, if finalized, will be on the RCRA Corrective Action Program. Specifically, when corrective action requirements are imposed at a RCRA treatment, storage, and disposal facility (TSDF), these specific PFAS would be among the hazardous constituents expressly identified for consideration in RCRA facility assessments and, where necessary, further investigation and cleanup through the RCRA corrective action process.

Some commenters suggest that the EPA failed to consider the costs and impacts of the proposed MCLs in non-drinking water contexts, such as its potential uses as CERCLA clean-up standards. As required by SDWA, this rule and analyses supporting the rulemaking only includes costs that ''are likely to occur solely as a result of compliance with the [MCL].'' (SDWA section 1412(b)(3)(C)(i)(III)) Thus, the EPA's cost analyses focused on the compliance costs of meeting the MCL to public water systems that are directly subject to this regulation. The same provision expressly directs the EPA to exclude ''costs resulting from compliance with other proposed or promulgated regulations.'' Thus, the EPA cannot consider the costs of use of the MCLs under other EPA statutes (such as CERCLA) as part of its EA because SDWA specifically excludes such consideration (42 U.S.C. 300g–1(b)(3)(C)(i)(III)). See also *City of Waukesha* v. *EPA,* 320 F.3d 228, 243–244 (D.C. Cir. 2003) (finding that SDWA excludes consideration of the costs of, for example, CERCLA compliance, as part of the required cost/benefit analysis). In addition, whether and how MCLs might be used in any particular clean-up is very site-specific and as a practical matter cannot be evaluated in this rule.

Many commenters compared the proposed MCLs to existing state and international standards, regulations, and guidelines. In particular, these commenters acknowledge the fact that several states have conducted their own rulemakings to promulgate MCLs and suggest that the EPA's analysis in support of the proposed MCLs are inconsistent with these state approaches. Further, these commenters ask the EPA to explain why certain states' cost-benefit analyses supported their respective levels and why the EPA's analysis is different. Regarding state PFAS regulations, the EPA disagrees with commenters who suggested that the agency should develop regulations consistent with current state-led actions in setting a national standard in accordance with SDWA. While some states have promulgated drinking water standards for various PFAS prior to promulgation of this NPDWR, this rule provides a nationwide, health protective level for PFOA and PFOS (as well as four other PFAS) in drinking water and reflects regulatory development requirements under SDWA, including the EPA's analysis of the best available and most recent peer-reviewed science; available drinking water occurrence, treatment, and analytical feasibility information relevant to the PQL; and consideration of costs and benefits. After the NPDWR takes effect, SDWA requires primacy states to have a standard that is no less stringent than the NPDWR. Additionally, analyses conducted by the agency in support of an NPDWR undergo a significant public engagement and peer review process. The EPA notes that the EA for this rule accounts for existing state standards at the time of analysis. Specifically, to estimate the costs and benefits of the final rule, the EPA assumed that occurrence estimates exceeding state limits are equivalent to the state-enacted limit. For these states, the EPA assumed that the state MCL is the maximum baseline PFAS occurrence value for all EP in the state. Additionally, while states may establish drinking water regulations or guidance values absent Federal regulation as they deem appropriate, the presence of state regulations does not preclude the EPA from setting Federal regulations under the authority of SDWA that meets that statute's requirements. For additional information on the EPA's EA, please see section XII.

3. Final Rule

After considering public comments, the EPA is finalizing enforceable MCLs for PFOA and PFOS at 4.0 ng/L as the closest feasible level to the MCLG. First, the agency is establishing non-enforceable MCLGs at zero for contaminants where no known or anticipated adverse effects to the health of persons will occur, allowing for an adequate margin of safety. The EPA then examined the treatment capability of BATs and the accuracy of analytical techniques as reflected in the PQL in establishing the closest feasible level. In evaluating feasibility, the agency has determined that multiple treatment technologies (*e.g.,* GAC, AIX) ''examined for efficacy under field conditions and not solely under laboratory conditions'' are found to be both effective and available to treat PFOA and PFOS to the standards and below. The EPA also determined that there are available analytical methods to measure PFOA and PFOS in drinking water and that the PQLs for both compounds reflect a level that can be achieved with sufficient precision and accuracy across laboratories nationwide using such methods. Since limits of analytical measurement for PFOA and PFOS require the MCL to be set at some level greater than the MCLG, the agency has determined that 4.0 ng/L (the PQL for each contaminant) represents the closest feasible level to the MCLG and the level at which laboratories using these methods can ensure, with sufficient accuracy and precision, that water systems nationwide can monitor and determine compliance so that they are ultimately delivering water that does not exceed the maximum permissible level of PFOA and PFOS to any user of their public water system. The EPA evaluates the availability and performance of BATs for treating water to minimize the presence of the contaminant consistent with the MCLG as well as the costs of applying those BATs to large metropolitan water systems when treating to that level. In consideration of these factors, the EPA is therefore establishing the MCL of 4.0 ng/L for both PFOA and PFOS. The EPA further notes that the agency has determined that the costs of SSCTs to reach 4.0 ng/L are affordable for households served by small drinking water systems. For additional discussion on the EPA's EA, please see section XII of this preamble. For additional discussion on the PQLs for the PFAS regulated as part of this NPDWR, please see section VII of this preamble. The EPA notes that upon consideration of information submitted by commenters regarding the implementation timeline for the rule, the agency is also exercising its authority under SDWA section 1412(b)(10) to allow two additional years for systems to comply with the MCL. For additional discussion on this extension, please see section XI.

The EPA clarifies that the MCLs for PFOA and PFOS are set using two significant digits in this final rule. In the proposed rule, the EPA proposed MCLGs for PFOA and PFOS at zero (0) and an enforceable MCL for PFOA and PFOS in drinking water with two significant digits at 4.0 ng/L. As previously discussed in section IV of this preamble, the MCLG for PFOA and PFOS is zero because these two PFAS are likely human carcinogens. Because the MCLGs are zero, the number of significant digits in the MCLGs are not the appropriate driver for considering the number of significant digits in the MCLs. This approach is consistent with other MCLs the EPA has set with carcinogenic contaminants, including for arsenic and bromate.

By setting the MCLs at 4.0, the EPA is setting the MCLs as close as feasible to the MCLGs. The EPA guidance states that all MCLs should be expressed in the number of significant digits permitted by the precision and accuracy of the specified analytical procedure(s) and that data reported should contain the same number of significant digits as the MCL (USEPA, 2000h). The EPA determined that two significant digits were appropriate for PFOA and PFOS considering existing analytical feasibility and methods. The EPA drinking water methods typically use two or three significant digits to determine concentrations. The EPA methods 533 and 537.1, those authorized for use in determining compliance with the MCLs, state that "[c]alculations must use all available digits of precision, but final reported concentrations should be rounded to an appropriate number of significant digits (one digit of uncertainty), typically two, and not more than three significant digits." The EPA has determined that both methods 533 and 537.1 provide sufficient analytical precision to allow for at least two significant digits.

B. PFAS Hazard Index: PFHxS, PFNA, HFPO–DA, and PFBS

1. Proposal

The EPA proposed an MCL for mixtures of PFHxS, PFNA, HFPO–DA, and PFBS expressed as a Hazard Index to protect against additive health concerns when present in mixtures in drinking water. As discussed in the March 2023 proposal (USEPA, 2023f), a Hazard Index is the sum of hazard quotients (HQs) from multiple substances. An HQ is the ratio of exposure to a substance and the level at which adverse effects are not anticipated to occur. The EPA proposed the MCL for mixtures of PFHxS, PFNA, HFPO–DA, and PFBS as the same as the MCLG: as proposed, the Hazard Index must be equal to or less than 1.0. This approach would set a permissible level for the contaminant mixture (*i.e.,* a resulting PFAS mixture Hazard Index greater than 1.0 is an exceedance of the health protective level and has potential human health risk for noncancer effects from the PFAS mixture in water). The proposal defined a mixture as containing one or more of the four PFAS and therefore covered each contaminant individually if only one of the four PFAS occurred. Thus, the Hazard Index as proposed ensures that the level of exposure to an individual PFAS remains below that which could impact human health because the exposure for that measured PFAS is divided by its corresponding HBWC. The EPA proposed HBWCs of 9.0 ng/L[5] for PFHxS; 10.0 ng/L for HFPO–DA; 10.0 ng/L for PFNA; and 2000.0 ng/L for PFBS (USEPA, 2023e).

The EPA requested comment on the feasibility of the proposed Hazard Index MCL, including analytical measurement and treatment capability, as well as reasonable costs, as defined by SDWA.

2. Summary of Major Public Comments and EPA Responses

The EPA received many comments supporting the use of the Hazard Index approach and regulation of additional PFAS. Consistent with these comments, through this action, the agency is establishing drinking water standards for PFHxS, PFNA, HFPO–DA, and PFBS (as well as PFOA and PFOS) to provide health protection against these contaminants found in drinking water. The EPA considered PFAS health effects information, evidence supporting dose additive health concerns from co-occurring PFAS, as well as national and state data for the levels of multiple PFAS in finished drinking water.

A few commenters disagreed with the EPA's feasibility evaluation in setting the MCL at the MCLG (*i.e.,* Hazard Index value of 1.0). Some of these commenters assert that technologies to remove the Hazard Index PFAS are not the same as those that effectively remove PFOA and PFOS. A couple of commenters were concerned that meeting the Hazard Index MCL may require more frequent media change-outs (*e.g.,* GAC), thereby increasing operating costs such that the Hazard Index MCL of 1.0 is not feasible. The agency disagrees with these commenters. As described above in part A of this section for PFOA and PFOS, the agency similarly considered feasibility as defined by SDWA for PFHxS, PFNA, HFPO–DA, and PFBS. First, the EPA established a Hazard Index MCLG as a Hazard Index of 1 for mixtures of PFHxS, PFNA, HFPO–DA, and PFBS. As part of setting the Hazard Index MCLG, the agency defined an HBWC for PFHxS, PFNA, HFPO–DA, and PFBS used in the calculation (see discussion in section IV of this preamble for further information).[6]

In considering the feasibility of setting the MCLs as close as feasible to the MCLG, the EPA first evaluated the (1) the availability of analytical methods to reliably quantify levels of the contaminants in drinking water and (2)

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[5] Some commenters noted an error in the HBWC calculation for PFHxS which was reported as 9.0 ng/L in the proposal. The agency has corrected the value in this NPDWR and within the requirements under 40 CFR part 141 subpart Z. The correct HRL/HBWC for PFHxS is 10 ng/L.

[6] The EPA notes that the HBWC are akin to an MCLG in that they reflect a level below which there are no known or anticipated adverse effects over a lifetime of exposure, including for sensitive populations and life stages, and allows for an adequate margin of safety.

the lowest levels at which contaminants can be reliably quantified within specific limits of precision and accuracy during routine laboratory operating conditions using the approved methods (*i.e.,* the PQLs). The EPA determined that there are available analytical methods approved (*i.e.,* Methods 533 and 537.1, version 2.0) to quantify levels below these HBWC levels. In addition, the PQLs for PFHxS, PFNA, HFPO–DA, and PFBS (between 3.0 to 5.0 ng/L) are all lower than the respective HBWCs used in setting the Hazard Index MCLG for each of these PFAS (10 ng/L for PFHxS, PFNA, and PFHxS, and 2000 ng/L for PFBS). Thus, the PQLs are not a limiting factor in determining the MCL. Second, the EPA evaluated the availability and performance of Best Available Technologies (BATs) for treating water to minimize the presence of these contaminants consistent with the MCLGs (see section X for additional discussion on BATs) as well as the costs of applying those BATs to large metropolitan water systems when treating to that level. The EPA has found the same technologies identified for PFOA and PFOS are also both available and have reliably demonstrated PFAS removal efficiencies that may exceed >99 percent and can achieve concentrations less than the proposed Hazard Index MCL for PFHxS, PFNA, HFPO–DA, and PFBS, and that the cost of applying those technologies is reasonable for large metropolitan water systems. As discussed above, for contaminants where the MCLG is higher than the PQL, the EPA sets the MCL at the MCLG if treatment is otherwise feasible because the PQL is not a limiting factor. In consideration of the availability of feasible treatment technologies, approved analytical methods to reliably quantify levels of the contaminants in drinking water, the EPA's cost analysis, and the fact that the PQLs are below the HBWCs used in setting the Hazard Index MCLG, the agency determines that setting the MCL at the same level as the MCLG for mixtures of PFHxS, PFNA, HFPO–DA and PFBS is feasible. Thus, the EPA is setting the Hazard Index MCL of 1 for mixtures of PFHxS, PFNA, HFPO–DA, and/or PFBS. For additional discussion and considerations surrounding BATs, please see section X.A of this preamble. For more information about the EPA's cost estimates, please see section XII of this preamble.

Many commenters support excluding PFOA and PFOS from the Hazard Index MCL. The EPA agrees with these commenters as there are analytical limitations that would complicate including PFOA and PFOS in the Hazard Index. As discussed in section IV of this preamble of the Hazard Index approach, individual PFAS hazard quotients (HQs) are calculated by dividing the measured concentration of each component PFAS in water (*e.g.,* expressed as ng/L) by the corresponding health-based water concentration (HBWC) for each component PFAS (*e.g.,* expressed as ng/L). The HBWC is akin to an MCLG in that they reflect a level below which there are no known or anticipated adverse effects over a lifetime of exposure, including for sensitive populations and life stages, and allows for an adequate margin of safety. Since PFOA and PFOS are likely human carcinogens, the MCLG (and if included in the Hazard Index, the HBWC) for each contaminant is zero. The only feasible way to represent PFOA and PFOS in the Hazard Index approach would be to only consider values for PFOA and PFOS at or above the PQL of 4.0 ng/L, however the level at which no known or anticipated adverse effects on the health of persons would occur is well below the PQL. As a result, any measured concentration above 4.0 ng/L for PFOA and PFOS would result in an exceedance of the Hazard Index MCL. The Hazard Index is intended to capture the aggregate risks of the Hazard Index PFAS when the monitored concentration is above the PQL but below the HBWC. These risks are not relevant to PFOA and PFOS given their PQLs. Because of the PQL considerations discussed in the preceding section V.A of this preamble, the EPA is not including PFOA and PFOS in the final rule Hazard Index. Therefore, the EPA is finalizing individual MCLs for PFOA and PFOS but not including these contaminants in the Hazard Index.

A few commenters provided feedback on the EPA's request for comment regarding the usage of significant figures to express the MCLs. See discussion on this issue in section IV of this preamble above. In summary, after considering public comment, the EPA agrees that one (1) significant digit is appropriate for the individual PFAS for PFHxS, PFNA and HFPO–DA (*i.e.,* 10 ng/L rather than 10.0 ng/L), and Hazard Index MCL (*i.e.,* 1 rather than 1.0).

Some commenters asked about inclusion of other PFAS in the Hazard Index in future revisions. The agency believes the Hazard Index approach can be an adaptive and flexible framework for considering additional PFAS. The EPA is required to review NPDWRs every six years and determine which, if any, need to be revised (*i.e.,* the Six-Year Review Process). The purpose of the review is to evaluate current information for regulated contaminants and to determine if there is any new information on health effects, treatment technologies, analytical methods, occurrence and exposure, implementation and/or other factors that provides a health or technical basis to support a regulatory revision that will improve or strengthen public health protection. This process allows the agency to consider these and other information as appropriate in deciding whether existing NPDWRs should be identified as candidates for revision as required by SDWA.

Many commenters compared the proposed MCLs to existing state and international standards, regulations, and guidelines. In particular, these commenters acknowledge that several states have conducted their own rulemakings to promulgate MCLs and suggest that the EPA's analysis in support of the proposed MCLs is inconsistent with these state approaches. Further, these commenters ask the EPA to explain why certain states' cost-benefit analyses supported their respective levels and why the EPA's analysis is different. Regarding state PFAS regulations, the EPA disagrees with commenters who suggested that the agency should not develop regulations different from state-led actions. SDWA mandates Federal regulation where the EPA determines that a contaminant meets the criteria for regulation under the statute. Moreover, the EPA's rule sets a national standard in accordance with SDWA for certain PFAS in drinking water that provides important protections for all Americans served by PWSs. Please see discussion above in part A under this section for consideration for existing state and international standards.

A few commenters suggest a need for effective data management systems to implement the Hazard Index. These commenters indicated that it will be challenging to implement the Hazard Index as proposed due to the tracking of multiple contaminants and automating these data into existing data management systems. For discussion on rule implementation issues, including primacy agency record keeping and reporting requirements, please see section XI of this preamble.

Some commenters raised concerns that the EPA did not consider a sufficient range of regulatory alternatives. For example, a few commenters contend that the EPA violated 1412(b)(3)(C)(i) of SDWA and the Unfunded Mandates Reform Act (UMRA) because the agency did not identify and consider what they deem a reasonable number of regulatory alternatives for PFHxS, PFNA, HFPO–DA and its ammonium salts, and PFBS. Specifically, these commenters cite that the EPA only considered a single HBWC and did not consider any alternatives to the Hazard Index MCL of 1 itself. The EPA disagrees with these commenters.

SDWA does not require the agency to consider any certain number of alternative MCLs or a range of alternatives. SDWA 1412(b)(3)(C)(i)(IV) only requires that in developing the HRRCA, the agency must consider the ''incremental costs and benefits associated with each alternative maximum contaminant level considered.'' Thus, the agency must conduct a cost-benefit analysis with each alternative MCL that is considered, if any. The EPA maintains that the proposed rule and regulatory alternatives considered at proposal met all requirements to consider alternatives. In the proposed rule, the EPA did not separately present changes in quantified costs and benefits for these approaches because the agency described that including individual MCLs in addition to the Hazard Index approach will be not change costs and benefits relative to the proposal (*i.e.,* the same number of systems will incur identical costs to the proposed option and the same benefits will be realized). For the final rule, the EPA has also estimated the marginal costs for the individual PFHxS, PFNA, and HFPO–DA MCLs in the absence of the Hazard Index (See chapter 5.1.3 and appendix N.4 of the EA for details). The EPA notes that the costs for the individual PFHxS, PFNA, and HFPO–DA MCLs have been considered in this final rule. For further discussion of how the EPA considered the costs of the five individual MCLs and the HI MCL, see section XII.A.4 of this preamble.

The EPA identified and analyzed a reasonable number of regulatory alternatives to determine the MCL requirement in the proposed rule as required by UMRA. UMRA's requirement to identify and consider a reasonable number of regulatory alternatives builds on the assessment of feasible alternatives required in E.O. 12866.[7] Specifically, as described in the proposed rule, the EPA considered an alternative approach to the one proposed that only used the Hazard Index MCL. The proposal took comment on establishing individual MCLs instead of and in addition to using a mixture-based approach for PFHxS, PFNA, HFPO–DA, and/or PFBS in mixtures. In that proposal, the EPA described how a traditional approach may be warranted should the EPA not finalize a regulatory determination for mixtures of these PFAS. Under this alternative, ''the proposed MCLG and MCL for PFHxS would be 9.0 ng/L; for HFPO–DA the MCLG and MCL would be 10.0 ng/L; for PFNA the MCLG and MCL would be 10.0 ng/L; and for PFBS the MCLG and MCL would be 2000.0 ng/L.'' The agency requested comment on these alternatives for PFHxS, PFNA, HFPO–DA, and PFBS and whether these individual MCLs instead of or in addition to the Hazard Index approach would change public health protection, improve clarity of the rule, or change costs. Additionally, the EPA considered alternative mixture-based approaches such as a target organ-specific Hazard Index (TOSHI) or relative potency factor (RPF) approach. The agency requested comment on these approaches. Based on the EPA's technical expertise, the agency determined that the Hazard Index is the most cost-effective and least burdensome alternative for purposes of UMRA because this approach for mixtures that achieves the objectives of the rule because of the level of protection afforded for the evaluation of chemicals with diverse (but in many cases shared) health endpoints. The EPA followed agency chemical mixture guidance (USEPA, 1986; USEPA, 1991b; USEPA, 2000a, which explain that when the Hazard Index value is greater than one (1) then risk is indicated (because exposure exceeds toxicity). The agency did not propose alternative Hazard Index values (*i.e.,* higher Hazard Index values) because the EPA determined that a Hazard Index MCL of 1 is feasible: multiple treatment technologies are available and are found effective to treat to or below the MCL; the costs of applying these technologies to large metropolitan water systems are reasonable; and there are analytical methods available to reliably quantify the four PFAS captured in the Hazard Index MCL. In addition, these alternative Hazard Index or mixture-based approaches would not provide sufficient protection against dose-additive health concerns from co-occurring PFAS. For example, a higher Hazard Index value (*e.g.,* Hazard Index equal to 2) allows for exposure to be greater than the toxicity and will not result in a sufficient health-protective standard that is close as feasible to the MCLG, which is a level at which there are no known or anticipated adverse effects on human health and allows for an adequate margin of safety. The EPA notes that commenters have not provided support justifying an alternative MCL standard for the Hazard Index. For additional discussion on UMRA, please see chapter 9 of USEPA (2024g).

3. Final Rule

Through this action, the EPA is promulgating the Hazard Index MCL for mixtures of two or more of PFHxS, PFNA, HFPO–DA and PFBS. The following equation provides the calculation of the PFHxS, PFNA, HFPO–DA, and PFBS Hazard Index MCL as finalized:

$$HI\ MCL = \left(\frac{[HFPO-DA_{ng/L}]}{[10\,ng/L]}\right) + \left(\frac{[PFBS_{ng/L}]}{[2000\,ng/L]}\right) + \left(\frac{[PFNA_{ng/L}]}{[10\,ng/L]}\right) + \left(\frac{[PFHxS_{ng/L}]}{[10\,ng/L]}\right)$$

Where:

$HFPO–DA_{water}$ = monitored concentration of HFPO–DA in ng/L;

$PFBS_{water}$ = monitored concentration of PFBS;

$PFNA_{water}$ = monitored concentration of PFNA and

$PFHxS_{water}$ = monitored concentration of PFHxS

The presence of PFBS can only trigger an MCL violation if it is present as part of a mixture with at least one of the other three PFAS (PFHxS, PFNA and

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[7] See OMB Memorandum M–95–09, Guidance for Implementing Title II of S.1.

HFPO–DA). As such, elevated PFBS concentrations that would normally cause a Hazard Index exceedance in isolation will not cause a violation if none of the other three PFAS are present in the mixture. The EPA is promulgating individual MCLs for PFHxS, PFNA, and HFPO–DA as well the Hazard Index MCL for mixtures of PFHxS, PFNA, HFPO–DA and PFBS concurrent with final regulatory determinations for these contaminants (please see section III of this preamble for additional discussion on the EPA's regulatory determinations).

The EPA has determined that it is feasible to set the MCL at the same level as the MCLG for mixtures of PFHxS, PFNA, HFPO–DA and PFBS as current BATs can remove each contaminant to a level equal to or below their respective HBWC. In addition, there are analytical methods available for these contaminants and the PQL for each contaminant is below the level established by the MCLG. The EPA also considered costs and determined that establishing a Hazard Index MCL of 1 is reasonable based on consideration of the costs to large metropolitan water systems. These considerations support a determination that a Hazard Index MCL of 1 for mixtures of two or more of PFHxS, PFNA, HFPO–DA and PFBS is feasible and therefore the EPA is setting the MCL at the same level as the MCLG. The EPA's MCL of 1 establish a ''maximum permissible level of contaminant in water'' because it is a limit for a mixture with PFAS components that must be met before the water enters the distribution system. Public water systems use their monitoring results as inputs into the Hazard Index equation to determine whether they are delivering water to any user that meets the MCL. For additional discussion regarding the derivation of the individual HBWCs and MCLGs, please see discussion in section III and IV of this preamble above.

C. Individual MCLs: PFHxS, PFNA and HFPO–DA

1. Proposal

As described in section V.B of this preamble above, the EPA proposed an MCL for mixtures of PFHxS, PFNA, HFPO–DA and PFBS based on a Hazard Index. The EPA proposed to address its preliminary regulatory determinations for PFHxS, PFNA, HFPO–DA, and/or PFBS and mixtures of these PFAS together through the Hazard Index approach. The proposal defined a mixture as containing one or more of the four PFAS and therefore covered each contaminant individually if only one of the four PFAS occurred. The EPA considered and took comment on establishing individual MCLGs and MCLs in lieu of or in addition to the Hazard Index approach for mixtures of PFHxS, PFNA, HFPO–DA, and/or PFBS.

2. Summary of Major Public Comments and EPA Responses

Commenters were mixed on the EPA's request for public comment on the establishment of stand-alone MCLs in lieu of or in addition to the Hazard Index MCL. Many of the comments were related to risk communications and messaging to consumers. While several commenters favored stand-alone MCLs in lieu of the Hazard Index to improve communications to their customers, several other commenters recommended stand-alone MCLs in addition to the Hazard Index MCL to achieve this purpose. Several commenters opposed individual MCLs for some or all of the PFAS because they believe it may complicate risk communication. After consideration of public comments, the EPA is addressing the final individual regulatory determination for PFHxS, HFPO–DA, and PFNA by promulgating individual MCLGs and NPDWRs for PFHxS, HFPO–DA, and PFNA. The EPA is addressing the final mixture regulatory determination by promulgating a Hazard Index MCLG and NPDWR for mixtures containing two or more of PFHxS, PFNA, HFPO–DA, and PFBS. This approach avoids confusion caused by the EPA's proposal that covered all the preliminary regulatory determinations in one Hazard Index standard. The EPA agrees that proper risk communication is an important focus for water systems and believes that finalizing individual MCLs for PFHxS, PFNA and HFPO–DA may help support risk communication as utilities and the public may be more familiar with this regulatory framework. At the same time, since those individual MCLs do not address additional risks from co-occurring PFAS, the EPA is finalizing a Hazard Index MCL to address dose additive health concerns associated with mixtures of two or more of PFHxS, PFNA, HFPO–DA, and PFBS that co-occur in drinking water. For additional discussion on the Hazard Index approach and other mixture-based approaches (*e.g.,* TOSHI), please see section IV of this preamble above.

3. Final Rule

The EPA is promulgating individual MCLs for PFHxS, PFNA and HFPO–DA at the same level as their respective MCLGs (which are equivalent to the HBWCs). The EPA is finalizing individual MCLs as follows: HFPO–DA MCL = 10 ng/L; PFHxS MCL = 10 ng/L; and PFNA MCL = 10 ng/L. The EPA is promulgating individual MCLs for PFHxS, PFNA, and HFPO–DA as well the Hazard Index MCL for mixtures of PFHxS, PFNA, HFPO–DA and PFBS concurrent with final determinations for these contaminants (please see section III of this preamble for additional discussion on the EPA's regulatory determinations).

The agency considered feasibility as defined by SDWA and the EPA's feasibility justification for these individual PFHxS, PFNA and HFPO–DA MCLs are the same and based on the same information as the Hazard Index MCL discussed in V.B above. The EPA further notes that the Hazard Index MCLG applies to the entire mixture but the EPA's technical justification for the underlying values (*i.e.,* HBWCs) are the same as the individual MCLGs in this rule. In summary, the EPA has determined that it is feasible to set the individual MCLs at the MCLGs for PFHxS, PFNA and HFPO–DA because current BATs can remove each contaminant to a level equal to or below their respective MCLGs. In addition, there are analytical methods available for these contaminants and the practical quantitation level (PQL) for each contaminant is below the level established by the MCLG. The EPA also considered costs and determined that establishing individual MCLs of 10 ng/L for PFHxS, PFNA, and HFPO–DA is reasonable based on consideration of the costs to large metropolitan water systems. These considerations support a determination that individual MCLs of 10 ng/L for PFHxS, PFNA, and HFPO–DA are feasible and therefore the EPA is setting the MCL at the same level as the MCLG. For additional discussion regarding the derivation of the individual HBWCs and MCLGs, please see section III and IV of this preamble above.

VI. Occurrence

The EPA relied on multiple data sources, including Unregulated Contaminant Monitoring Rule (UCMR) 3 and state finished water data, to evaluate the occurrence of PFOA, PFOS, PFHxS, PFNA, and HFPO–DA and probability of co-occurrence of these PFAS and PFBS. The EPA also incorporated both the UCMR 3 and some state data into a Bayesian hierarchical model which supported exposure estimates for select PFAS at lower levels than were measured under UCMR 3. The EPA has utilized similar statistical approaches in past regulatory actions to inform its decision making, particularly where a contaminant's occurrence is at low concentrations

(USEPA, 2006c). The specific modeling framework used to inform this regulatory action is based on the peer-reviewed model published in Cadwallader et al. (2022). Collectively, these data and the occurrence model informed estimates of the number of water systems (and associated population) expected to be exposed to levels of the final and proposed alternative MCLs for PFOA and PFOS, the final MCLs for PFHxS, PFNA, and HFPO–DA, and the final Hazard Index MCL for PFHxS, PFNA, HFPO–DA, and PFBS.

The EPA notes that, as described in sections III and V of this preamble, the EPA is finalizing individual Maximum Contaminant Levels (MCLs) for three of the four Hazard Index PFAS (PFHxS, PFNA, and HFPO–DA) at 10 ng/L each. An analysis of occurrence relative to HRLs for PFHxS, PFNA, and HFPO–DA (which are the same as the final individual MCLs for these compounds at 10 ng/L) using UCMR 3 data and updated state datasets is presented in section III.C of this preamble and further described in the *Occurrence Technical Support Document* (USEPA, 2024b). The information in the following sections supports the agency's finding that PFHxS, PFNA, and HFPO–DA occur at a frequency and level of public health concern as discussed in section III.C of this preamble.

A. UCMR 3

1. Proposal

UCMR 3 monitoring occurred between 2013 and 2015 and is currently the best nationally representative finished water dataset for any PFAS, including PFOA, PFOS, PFHxS, PFNA, and PFBS. Under UCMR 3, 36,972 samples from 4,920 public water systems (PWSs) were analyzed for these five PFAS. PFOA was found above the UCMR 3 minimum reporting level (20 ng/L) in 379 samples at 117 systems serving a population of approximately 7.6 million people located in 28 states, Tribes, or U.S. territories. PFOS was found in 292 samples at 95 systems above the UCMR 3 minimum reporting level (40 ng/L). These systems serve a population of approximately 10.4 million people located in 28 states, Tribes, or U.S. territories. PFHxS was found above the UCMR 3 minimum reporting level (30 ng/L) in 207 samples at 55 systems that serve a population of approximately 5.7 million located in 25 states, Tribes, and U.S. territories. PFBS was found in 19 samples at 8 systems above the UCMR 3 minimum reporting level (90 ng/L). These systems serve a population of approximately 350,000 people located in 5 states, Tribes, and U.S. territories. Lastly, PFNA was found above the UCMR 3 minimum reporting level (20 ng/L) in 19 samples at 14 systems serving a population of approximately 526,000 people located in 7 states, Tribes, and U.S. territories.

2. Summary of Major Public Comments and EPA Responses

Some commenters supported the EPA's use of the best available public health information including data from UCMR 3 and state occurrence data. A few commenters criticized the use of UCMR 3 data, stating that the data suffer from limitations. These commenters expressed concern over the high minimum reporting levels, the exclusion of many small systems, and the lack of national monitoring of HFPO–DA. Some of these commenters assert that UCMR 3 does not represent best available occurrence data for this rule. The EPA disagrees with these commenters. While UCMR 3 does have higher reporting limits than those available through current analytical methods, the data still provides the best available nationwide occurrence data to inform the occurrence and co-occurrence profile for the regulated PFAS for which monitoring was conducted. These data are also a critical component of the EPA's model to estimate national level occurrence for certain PFAS and ensure it is nationally representative (see subsection E of this section). The EPA also disagrees that the UCMR 3 excludes small water systems as it included a statistically selected, nationally representative sample of 800 small drinking water systems. Regarding commenter concerns for lack of UCMR monitoring data on HFPO–DA, the agency notes that the EPA examined recent data collected by states who have made their data publicly available. A discussion of these data and public comments on this information is presented in sections III.C and VI.B of this preamble.

3. Final Rule

After considering public comment, the EPA maintains that UCMR 3 data are the best available, complete nationally representative dataset and they play an important role in supporting the EPA's national occurrence analyses, demonstrating occurrence and co-occurrence of the monitored PFAS in drinking water systems across the country that serve millions of people.

B. State Drinking Water Data

1. Proposal

The agency has supplemented the UCMR 3 data with more recent data collected by states who have made their data publicly available. In general, the large majority of these more recent state data were collected using newer EPA-approved analytical methods and state results reflect lower reporting limits than those in the UCMR 3. State results show continued occurrence of PFOA, PFOS, PFHxS, PFNA, and PFBS in multiple geographic locations. These data also show these PFAS occur at lower concentrations and significantly greater frequencies than were measured under the UCMR 3 (likely because the more recent monitoring was able to rely on more sensitive analytical methods). Furthermore, these state data include results for more PFAS than were included in the UCMR 3, including HFPO–DA.

At the time of proposal, the EPA evaluated publicly available state monitoring data from 23 states, representing sampling conducted on or before May 2021. The EPA acknowledged that the available data were collected under varying circumstances; for example, targeted vs. non-targeted monitoring (*i.e.,* monitoring not conducted specifically in areas of known or potential contamination). Due to the variability in data quality, the EPA further refined this dataset based on representativeness and reporting limitations, resulting in detailed technical analyses using a subset of the available state data. A comprehensive discussion of all the available state PFAS drinking water occurrence data was included in the *Occurrence Technical Support Document* (USEPA, 2023l).

2. Summary of Major Public Comments and EPA Responses

Commenters generally supported the use of state datasets. A few commenters discussed their own PFAS occurrence data, some of which were provided to the EPA, relative to the EPA's proposed regulatory levels and/or provided summaries of other monitoring efforts. Where possible, the EPA presents this information within its occurrence analysis—see the *Other Data* sections of USEPA (2024b). A few commenters recommended that the EPA expand the datasets used for the final rule to include additional and updated state sampling information. The EPA agrees with these suggestions to rely on additional and updated sampling information in order to evaluate PFAS occurrence in drinking water. Therefore, the agency has included updated information in its occurrence analyses as described in section VI.B.3 of this preamble. The EPA notes that this information is consistent with the analyses contained in the proposal for this action.

A few commenters criticized the use of state datasets in occurrence analyses. These commenters claimed that the state datasets were insufficient for national extrapolation and not dependable due to being collected under variable circumstances. These commenters expressed the need for enhanced quality control (QC) by the EPA to exclude data below reasonable reporting thresholds. The agency disagrees with commenters who contend that state datasets are insufficient for national extrapolation. For both the rule proposal and this final action, the EPA took QC measures to ensure the EPA used the best available data for national extrapolation. For example, the EPA acknowledged in the proposal that states used various reporting thresholds when presenting their data, and for some states there were no clearly defined reporting limits. The EPA identified state reporting thresholds where possible and, when appropriate, incorporated individual state-specific thresholds when conducting data analyses. For other states, the EPA presented the data as provided by the state. Due to the reporting limitations of some of the available state data (*e.g.,* reporting combined analyte results rather than individual analyte results), the EPA did not utilize all of these data in the subsequent occurrence analyses/co-occurrence analyses. Specific data analysis criteria (*e.g.,* separation of non-targeted and targeted monitoring results) were also applied. Additionally, the agency also verified that the vast majority of the data were collected using EPA-approved methods. Further, the EPA reviewed all available data thoroughly to ensure that only finished drinking water data were presented. A description of the scope and representativeness of the state data was provided in the proposal of this action in the *PFAS Occurrence and Contaminant Background Support Document* (USEPA, 2023l). These include describing the states the EPA found to have publicly available data, identifying the reporting thresholds where possible, and distinguishing whether monitoring was non-targeted or targeted (*i.e.,* monitoring in areas of known or potential PFAS contamination). These QC measures ensured that the EPA utilized the best available data for national extrapolation.

3. Final Rule

In the proposed rule preamble, the EPA discussed how states may have updated data available and that additional states have or intend to conduct monitoring of finished drinking water and that the agency would consider these additional data to inform this final regulatory action. After consideration of all the public comments on this issue, the EPA has updated its analysis of state monitoring data by including results that were available as of May 2023. This updated state dataset includes publicly available data from 32 states: Alabama, Arizona, California, Colorado, Delaware, Georgia, Idaho, Illinois, Indiana, Iowa, Kentucky, Maine, Maryland, Massachusetts, Michigan, Minnesota, Missouri, New Hampshire, New Jersey, New Mexico, New York, North Carolina, North Dakota, Ohio, Oregon, Pennsylvania, South Carolina, Tennessee, Vermont, Virginia, West Virginia, and Wisconsin. The dataset includes data from 9 states that were not available at the time of proposal.

Tables 4 and 5 in this section demonstrate the number and percent of samples with PFOA and PFOS based on state-reported detections, and the number and percent of systems with PFOA and PFOS based on state-reported detections, respectively, for the non-targeted state finished water monitoring data. Section III.B. of this preamble describes the state reported finished water occurrence data for PFHxS, PFNA, HFPO–DA, and PFBS data.

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Table 4. Non-Targeted State PFOS and PFOA Finished Water Data – Summary of Samples with State Reported Detections[1]

State	**PFOS state reported sample detections**	**PFOS state reported sample detection (percent)**	**PFOA state reported sample detections**	**PFOA state reported sample detections (percent)**
Alabama[2]	249	N/A	176	N/A
Colorado	60	10.3%	54	9.3%
Illinois	306	14.3%	298	14.0%
Indiana	8	1.7%	8	1.7%
Kentucky	33	40.7%	24	29.6%
Maine	101	14.3%	142	20.1%
Maryland	17	19.3%	20	22.7%
Massachusetts	4432	47.4%	5363	57.4%
Michigan	489	4.6%	557	5.2%
Missouri	22	9.2%	17	7.1%
New Hampshire	495	27.3%	1010	55.7%
New Jersey	6502	40.9%	8063	50.7%
New York	1576	22.3%	1751	24.8%
North Dakota	3	2.6%	2	1.7%
Ohio	113	5.8%	116	6.0%
South Carolina	135	17.6%	141	18.3%
Tennessee	0	0.0%	0	0.0%
Vermont	192	12.3%	225	14.4%
Wisconsin	187	23.9%	167	21.2%

Notes:

[1] Detections determined by individual state reported limits which are not defined consistently across all states.

[2] State only reported detections (i.e., there was no information on total number of samples collected)

Table 5: Non-Targeted State PFOS and PFOA Finished Water Data – Summary of Monitored Systems with State Reported Detections[1]

State	PFOS Monitored Systems with State Reported Detections	PFOS Monitored Systems with State Reported Detections (Percent)	PFOA Monitored Systems with State Reported Detections	PFOA Monitored Systems with State Reported Detections (Percent)
Alabama[2]	88	N/A	65	N/A
Colorado	50	12.6%	45	11.3%
Illinois	73	7.3%	67	6.7%
Indiana	7	1.9%	8	2.2%
Kentucky	30	40.5%	22	29.7%
Maine	94	14.6%	132	20.4%
Maryland	9	14.3%	10	15.9%
Massachusetts	417	31.4%	520	39.1%
Michigan	105	4.2%	135	5.4%
Minnesota	55	9.5%	69	12.0%
Missouri	11	8.8%	7	5.6%
New Hampshire	189	33.8%	310	55.4%
New Jersey	541	48.2%	625	55.7%
New York	496	26.3%	558	29.6%
North Dakota	6	5.4%	7	6.3%
Ohio	29	2.0%	33	2.2%
South Carolina	80	26.7%	85	28.3%
Tennessee	0	0.0%	0	0.0%
Vermont	38	6.7%	49	8.7%
Wisconsin	70	29.3%	66	27.6%

Notes:

[1] Detections determined by individual state reported limits which are not defined consistently across all states.

[2] State only reported detections (i.e., there was no information on total number of samples collected)

As illustrated in Tables 4 and 5, there is a wide range in PFOA and PFOS results between states. Nonetheless, more than one-third of states that conducted non-targeted monitoring observed PFOA and/or PFOS at more than 25 percent of systems. Among the detections, PFOA concentrations ranged from 0.21 to 650 ng/L with a range of median concentrations from 1.27 to 5.61 ng/L, and PFOS concentrations ranged from 0.24 to 650 ng/L with a range of median concentrations from 1.21 to 12.1 ng/L.

Monitoring data for PFOA and PFOS from states that conducted targeted monitoring efforts, including 15 states, demonstrate results consistent with the non-targeted state monitoring. For example, in Pennsylvania, 26.3 and 24.9 percent of monitored systems found PFOA and PFOS, respectively, with reported concentrations of PFOA ranging from 1.7 to 59.6 ng/L and PFOS ranging from 1.8 to 94 ng/L. California reported 35.8 and 39.0 percent of monitored systems found PFOA and PFOS, respectively, including reported concentrations of PFOA ranging from 0.9 to 190 ng/L and reported concentrations of PFOS from 0.4 to 250 ng/L. In Maryland, PFOA and PFOS were found in 57.6 and 39.4 percent of systems monitored, respectively, with reported concentrations of PFOA ranging from 1.02 to 23.98 ng/L and reported concentrations of PFOS ranging from 2.05 to 235 ng/L. In Iowa, PFOA and PFOS were found in 11.2 and 12.1 percent of systems monitored, respectively, with reported concentrations of PFOA ranging from 2 to 32 ng/L and reported concentrations of PFOS ranging from 2 to 59 ng/L.

As discussed above in section V of this preamble, the EPA is finalizing individual MCLs of 4.0 ng/L for PFOA and PFOS, individual MCLs for PFHxS, PFNA, and HFPO–DA, and a Hazard Index level of 1 for PFHxS, PFNA, HFPO–DA, and PFBS. The EPA also evaluated occurrence for the regulatory alternatives discussed in section V of this preamble, including alternative MCLs for PFOA and PFOS of 5.0 ng/L and 10.0 ng/L. Table 6, Table 7, and Table 8 demonstrate, based on available state data, the total reported number and percentages of monitored systems that exceed these proposed and alternative MCL values across the non-targeted state finished water monitoring data.

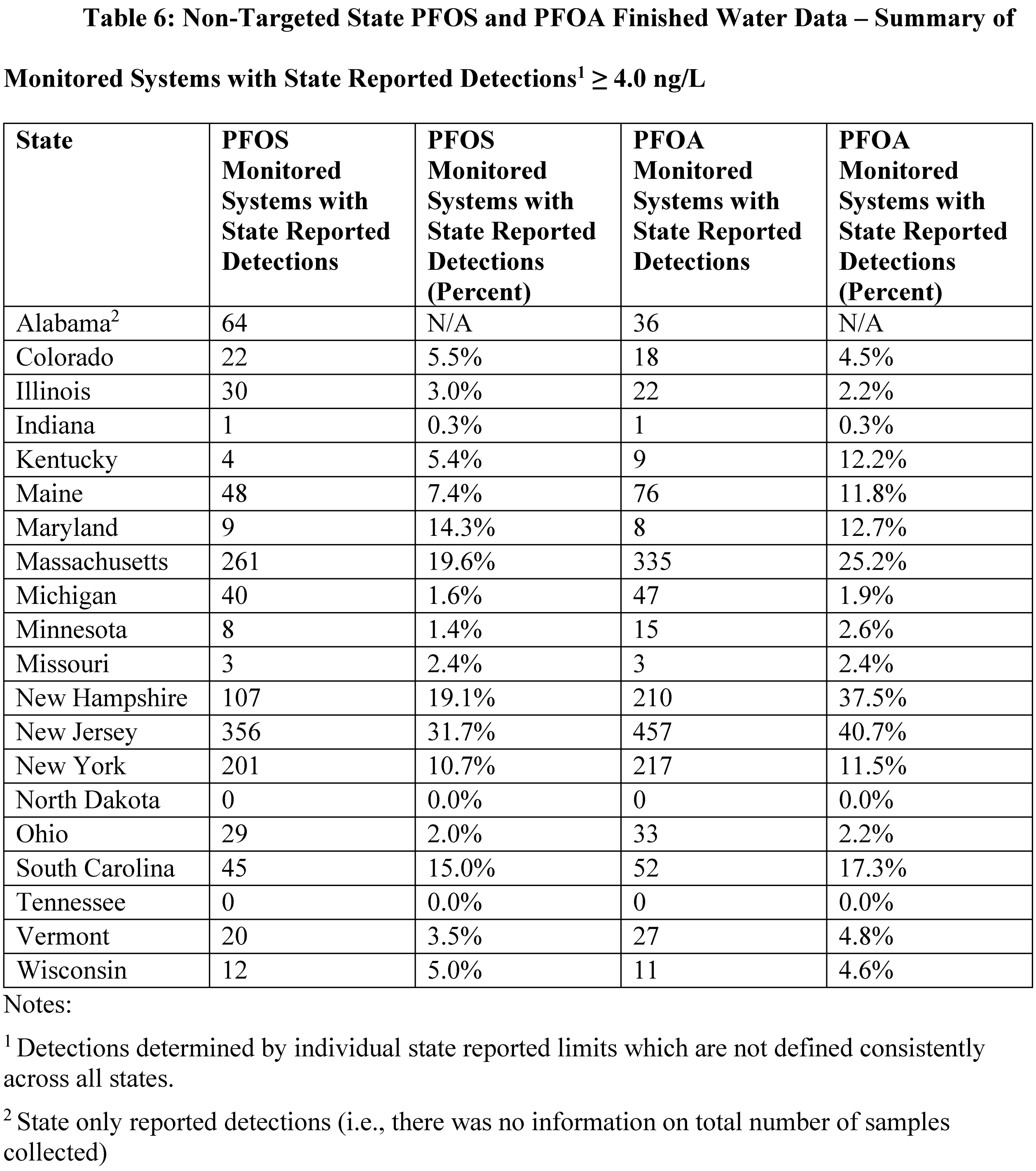

Table 6: Non-Targeted State PFOS and PFOA Finished Water Data – Summary of Monitored Systems with State Reported Detections[1] ≥ 4.0 ng/L

State	PFOS Monitored Systems with State Reported Detections	PFOS Monitored Systems with State Reported Detections (Percent)	PFOA Monitored Systems with State Reported Detections	PFOA Monitored Systems with State Reported Detections (Percent)
Alabama[2]	64	N/A	36	N/A
Colorado	22	5.5%	18	4.5%
Illinois	30	3.0%	22	2.2%
Indiana	1	0.3%	1	0.3%
Kentucky	4	5.4%	9	12.2%
Maine	48	7.4%	76	11.8%
Maryland	9	14.3%	8	12.7%
Massachusetts	261	19.6%	335	25.2%
Michigan	40	1.6%	47	1.9%
Minnesota	8	1.4%	15	2.6%
Missouri	3	2.4%	3	2.4%
New Hampshire	107	19.1%	210	37.5%
New Jersey	356	31.7%	457	40.7%
New York	201	10.7%	217	11.5%
North Dakota	0	0.0%	0	0.0%
Ohio	29	2.0%	33	2.2%
South Carolina	45	15.0%	52	17.3%
Tennessee	0	0.0%	0	0.0%
Vermont	20	3.5%	27	4.8%
Wisconsin	12	5.0%	11	4.6%

Notes:

[1] Detections determined by individual state reported limits which are not defined consistently across all states.

[2] State only reported detections (i.e., there was no information on total number of samples collected)

Table 7: Non-Targeted State PFOS and PFOA Finished Water Data – Summary of Monitored Systems with State Reported Detections[1] ≥ 5.0 ng/L

State	PFOS Monitored Systems with State Reported Detections	PFOS Monitored Systems with State Reported Detections (Percent)	PFOA Monitored Systems with State Reported Detections	PFOA Monitored Systems with State Reported Detections (Percent)
Alabama[2]	53	N/A	30	N/A
Colorado	16	4.0%	14	3.5%
Illinois	23	2.3%	13	1.3%
Indiana	1	0.3%	1	0.3%
Kentucky	3	4.1%	4	5.4%
Maine	38	5.9%	67	10.4%
Maryland	5	7.9%	8	12.7%
Massachusetts	220	16.5%	280	21.0%
Michigan	36	1.4%	35	1.4%
Minnesota	7	1.2%	12	2.1%
Missouri	2	1.6%	3	2.4%
New Hampshire	86	15.4%	186	33.2%
New Jersey	306	27.2%	409	36.4%
New York	154	8.2%	183	9.7%
North Dakota	0	0.0%	0	0.0%
Ohio	29	2.0%	33	2.2%
South Carolina	36	12.0%	38	12.7%
Tennessee	0	0.0%	0	0.0%
Vermont	16	2.8%	23	4.1%
Wisconsin	10	4.2%	5	2.1%

Notes:

[1] Detections determined by individual state reported limits which are not defined consistently across all states.

[2] State only reported detections (i.e., there was no information on total number of samples collected)

Table 8: Non-Targeted State PFOS and PFOA Finished Water Data – Summary of Monitored Systems with State Reported Detections[1] ≥ 10.0 ng/L

State	PFOS Monitored Systems with State Reported Detections	PFOS Monitored Systems with State Reported Detections (Percent)	PFOA Monitored Systems with State Reported Detections	PFOA Monitored Systems with State Reported Detections (Percent)
Alabama[2]	34	N/A	18	N/A
Colorado	3	0.8%	2	0.5%
Illinois	5	0.5%	7	0.7%
Indiana	0	0.0%	0	0.0%
Kentucky	1	1.4%	1	1.4%
Maine	10	1.5%	32	5.0%
Maryland	5	7.9%	7	11.1%
Massachusetts	112	8.4%	123	9.2%
Michigan	16	0.6%	17	0.7%
Minnesota	2	0.3%	4	0.7%
Missouri	0	0.0%	1	0.8%
New Hampshire	39	7.0%	83	14.8%
New Jersey	159	14.2%	223	19.9%
New York	57	3.0%	64	3.4%
North Dakota	0	0.0%	0	0.0%
Ohio	21	1.4%	15	1.0%
South Carolina	12	4.0%	8	2.7%
Tennessee	0	0.0%	0	0.0%
Vermont	7	1.2%	7	1.2%
Wisconsin	8	3.3%	0	0.0%

Notes:

[1] Detections determined by individual state reported limits which are not defined consistently across all states.

[2] State only reported detections (i.e., there was no information on total number of samples collected)

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Based on the available state data presented in Table 6, Table 7, and Table 8, within 20 states that conducted non-targeted monitoring there are 1,260 systems with results above the PFOS MCL of 4.0 ng/L and 1,577 systems with results above the PFOA MCL of 4.0 ng/L. These systems serve populations of 12.5 and 14.4 million people, respectively. As expected, the number of systems exceeding either of the proposed alternative MCLs decreases as the values are higher; however, even at the highest alternative PFOS and PFOA MCL values of 10.0 ng/L, there are still 491 and 612 systems with exceedances, serving populations of approximately 5.3 and 6.0 million people, respectively.

Monitoring data for PFOA and PFOS from states that conducted targeted sampling efforts shows additional systems that would exceed the final and alternative MCLs. For example, in California, Maine, Maryland, and Pennsylvania, 30.9 percent (38 PWSs), 27.8 percent (5 PWSs), 25 percent (18 PWSs), and 19.3 percent (66 PWSs) of monitored systems reported results above the proposed PFOS MCL of 4.0 ng/L, respectively, and 29.3 percent (36 PWSs), 27.8 percent (5 PWSs), 25 percent (18 PWSs), and 21.1 percent (72 PWSs) of monitored systems reported results above the proposed PFOA MCL of 4.0 ng/L, respectively. While these frequencies may be anticipated given the sampling locations, within only these four states that conducted limited, targeted monitoring, the monitored systems with results above the proposed PFOS MCL and proposed PFOA MCL serve significant populations of approximately 5.7 million people and approximately 5.6 million people, respectively.

C. PFAS Co-Occurrence

While the discussions in sections III.B, VI.A. and VI.B of this preamble describe how PFOA, PFOS, PFHxS, PFNA, and HFPO–DA occur individually, numerous studies and analyses have documented that PFAS co-occur in finished drinking water (Adamson et al., 2017; Cadwallader et al., 2022; Guelfo and Adamson, 2018). As discussed in section V of this preamble, the EPA is finalizing regulation of mixtures that include at least two of PFHxS, PFNA, HFPO–DA, and PFBS (collectively referred to as ''Hazard Index PFAS'') as part of a Hazard Index approach.

1. Proposal

In the March 2023 proposal preamble, the EPA presented occurrence data that illustrated the extent to which PFOA, PFOS, PFHxS, PFNA, HFPO–DA, and PFBS co-occur in drinking water. Co-occurrence analyses primarily utilized available non-targeted state PFAS finished drinking water data, though UCMR 3 data analysis is presented in the *PFAS Occurrence and Contaminant Background Support Document* (USEPA, 2024b). The EPA also conducted two separate analyses using state datasets to determine the extent to which these six PFAS co-occur: a groupwise analysis and a pairwise analysis.

When analyzing PFAS co-occurrence, groupwise analysis is important for determining whether the presence of PFOA and PFOS provides insight regarding the likelihood of Hazard Index PFAS being present as well, which has broad implications for public health. This is because occurrence information for the Hazard Index PFAS is less extensive than the occurrence information for PFOA and PFOS due to fewer states monitoring the Hazard Index PFAS; therefore, establishing co-occurrence with PFOA and PFOS helps with understanding the extent of general Hazard Index PFAS occurrence. For the groupwise analysis, the six PFAS were separated into two groups—one consisted of PFOS and PFOA and the other group included the four Hazard Index PFAS. The analysis broke down the systems and samples according to whether chemicals from the respective groups were detected. Results were also shown separated by state. Results generally indicated that when PFOA or PFOS were found, Hazard Index PFAS were considerably more likely to also be found. This implies that, for systems that only measured PFOA and/or PFOS, detected those PFAS, and did not measure the Hazard Index PFAS, the Hazard Index PFAS are more likely to also be present than if PFOA and/or PFOS were not detected. At a national level, since many systems monitored for PFOA and PFOS only and detected these PFAS, this means that estimates of Hazard Index PFAS occurrence based on state Hazard Index PFAS data alone are likely to be underestimated. Given that the state datasets varied in the specific PFAS that were monitored, the analysis also compared the number of Hazard Index PFAS analyzed with the number of Hazard Index PFAS reported present. As more Hazard Index PFAS were analyzed, more Hazard Index PFAS were found. Further, systems and samples where Hazard Index PFAS were found were more likely to find multiple Hazard Index PFAS than a single Hazard Index PFAS (when monitoring for 3 or 4 Hazard Index PFAS).

Given that the groupwise co-occurrence analysis established that the Hazard Index PFAS, as a group, occur with a substantial level of frequency, particularly alongside PFOA or PFOS, the pairwise co-occurrence is relevant for understanding how the individual PFAS included in the rule co-occur with each other. The pairwise co-occurrence analysis explored the odds ratios for each unique pair of PFAS included in the regulation. Pairwise co-occurrence through odds ratios showed statistically significant relationships between nearly all unique pairs of PFAS included in the proposed rule. Odds ratios reflect the change in the odds of finding one chemical (*e.g.,* Chemical A) given that the second chemical (*e.g.,* Chemical B) is known to be present compared to the odds of finding it if the second chemical is not present. For example, an odds ratio of 2 would indicate that the presence of the second chemical would be expected to double the odds of the first chemical being reported present. An odds ratio of 1 indicates that there is no association between the two chemicals. At the system level, point odds ratios estimates ranged from 1.7–142.7, indicating that in some instances the odds of finding one PFAS increased by more than two orders of magnitude if the other PFAS was reported present (in other words, for some PFAS combinations, if one PFAS is present, there is more than 100 times the odds of certain other PFAS being present). HFPO–DA and PFHxS was the only pair of PFAS chemicals included in the proposed regulation that did not have a statistically significant relationship; 1 fell within the 95 percent confidence interval, indicating that the odds ratio was not determined to be statistically significantly different from 1.

In the proposed rule, the agency determined that, both as a group and as individual chemicals, the Hazard Index PFAS had a higher likelihood of being reported if PFOS or PFOA were present, First, the groupwise analysis established that the Hazard Index PFAS, in addition to PFOA and PFOS, occur at a significant frequency in drinking water. Then, the pairwise analysis demonstrated that PFOA, PFOS, PFHxS, PFNA, HFPO–DA, and PFBS (the individual PFAS) generally co-occur with each other, as opposed to occurring independently. These data further support the EPA's finding that these PFAS are likely to occur, and that there is a substantial likelihood that combinations of PFHxS, PFNA, HFPO–DA, and PFBS co-occur in mixtures with a frequency of public health concern in drinking water systems.

2. Summary of Major Public Comments and EPA Responses

Some commenters agreed with the agency's conclusion in the March 2023 proposal that the PFAS included in the regulation appeared to meaningfully co-occur. However, some other commenters stated that they believed the data used to assess PFAS co-occurrence were too limited to make substantive conclusions. The EPA disagrees that the data were too limited or that the co-occurrence analysis was inconclusive. Based on the non-targeted state monitoring data used in the co-occurrence analysis (from 11 states), findings of the pairwise and groupwise analyses established a strong likelihood that these chemicals meaningfully co-occur in drinking water. This was observed through odds ratios statistically significantly greater than 1 in the pairwise analysis as well as frequency at which multiple chemicals were detected in the groupwise analysis. Based on public comment, the agency has updated its analysis to include more recent non-targeted state data that became publicly available after the proposal analyses were finalized. This ensures that findings are up to date; as discussed further in the following subsection, the more recent data confirms the proposal analysis.

3. Final Rule

After considering public comment and updating analyses, the EPA concluded that the co-occurrence analyses continue to support the premise in the proposed rule that PFAS are likely to co-occur and support the EPA's final rule approach. Following is a discussion and presentation of information related to the EPA's co-occurrence analysis for this final rule effort. These data include all data from the rule proposal, in addition to the updated data the EPA incorporated based on public comment. As discussed elsewhere in this preamble, the newer data confirm the EPA's conclusions from proposal.

a. Groupwise Chemical Co-Occurrence

Table 9 shows the distribution of systems and samples according to whether states reported detections for any Hazard Index PFAS (PFHxS, PFNA, HFPO–DA, and PFBS) and whether they also reported detections of PFOS or PFOA. USEPA (2024b) provides additional information for this analysis.

Table 9: Non-Targeted State PFAS Finished Water Data – Samples and Systems Binned According to Whether PFOS or PFOA were Reported by States and Whether Additional Hazard Index PFAS were Reported

Type	No PFOS or PFOA Reported		PFOS or PFOA Reported		Total Count
	No HI PFAS Reported	At Least One HI PFAS Reported	No HI PFAS Reported	At Least One HI PFAS Reported	
Samples	28,249 (57.8%)	1,321 (2.7%)	7,365 (15.1%)	11,954 (24.5%)	48,889
Systems	8,576 (70.6%)	401 (3.3%)	1,079 (8.9%)	2,089 (17.2%)	12,145

Considering eligible samples and systems within the aggregated state dataset, states reported either PFOA, PFOS, or one or more Hazard Index PFAS in 42.2 percent (20,640 of 48,889) of samples and 29.4 percent (3,569 of 12,145) of systems. When any PFAS (among PFOA, PFOS, and the Hazard Index PFAS) were reported, at least one Hazard Index PFAS was also reported in 64.3 percent (13,275 of 20,640) of samples and at 69.8 percent (2,490 of 3,569) of systems. Further, among samples and systems that reported PFOS or PFOA, at least one Hazard Index PFAS was reported in 61.9 percent (11,954 of 19,319) of samples and at 65.9 percent (2,089 of 3,168) of systems. This demonstrated strong co-occurrence of Hazard Index PFAS with PFOA and PFOS and a substantial likelihood (over 60 percent) of at least one Hazard Index PFAS being present at systems reporting the presence of PFOS or PFOA. Overall, one or more Hazard Index PFAS were reported at about 20.5 percent (2,490 of 12,145) of systems included in the aggregated state dataset of non-targeted monitoring. If this percentage were extrapolated to the nation, one or more Hazard Index PFAS would be found in over 13,000 systems. Table 10 shows the distribution of systems in a similar manner but provides a breakdown by state and includes only systems that monitored for either three or four of the Hazard Index PFAS.

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Table 10: Non-Targeted State PFAS Finished Water Data – Systems that Sampled for 3 or 4 Hazard Index PFAS Binned According to Whether PFOS or PFOA were Reported and Whether Any Additional Hazard Index PFAS were Reported by State

State	No PFOA/S Reported		PFOA/S Reported		Total System Count
	No HI Reported	HI Reported	No HI Reported	HI Reported	
CO	270 (68.0%)	26 (6.5%)	11 (2.8%)	90 (22.7%)	397
IL	880 (88.4%)	28 (2.8%)	25 (2.5%)	63 (6.3%)	996
IN	339 (91.4%)	19 (5.1%)	6 (1.6%)	7 (1.9%)	371
KY	38 (51.4%)	3 (4.1%)	17 (23.0%)	16 (21.6%)	74
MA	479 (36.5%)	33 (2.5%)	146 (11.1%)	655 (49.9%)	1,313
MD	51 (81.0%)	0 (0.0%)	3 (4.8%)	9 (14.3%)	63
ME	469 (73.2%)	12 (1.9%)	84 (13.1%)	76 (11.9%)	641
MI	2,205 (87.9%)	130 (5.2%)	66 (2.6%)	107 (4.3%)	2,508
MO	102 (90.3%)	2 (1.8%)	4 (3.5%)	5 (4.4%)	113
ND	99 (89.2%)	9 (8.1%)	0 (0.0%)	3 (2.7%)	111
NH	64 (27.0%)	13 (5.5%)	68 (28.7%)	92 (38.8%)	237
NJ	227 (34.1%)	7 (1.1%)	142 (21.4%)	289 (43.5%)	665
NY	275 (40.1%)	15 (2.2%)	132 (19.2%)	264 (38.5%)	686
OH	1,397 (94.5%)	31 (2.1%)	25 (1.7%)	26 (1.8%)	1,479
SC	187 (62.8%)	11 (3.7%)	28 (9.4%)	72 (24.2%)	298
TN	1 (100.0%)	0 (0.0%)	0 (0.0%)	0 (0.0%)	1
VT	492 (87.2%)	14 (2.5%)	26 (4.6%	32 (5.7%)	564
WI	140 (60.1%)	24 (10.3%)	10 (4.3%)	59 (25.3%)	233

Tennessee only had data from one system which did not report the presence of any of the six PFAS. Otherwise, the percentage of systems included in Table 10 that reported any Hazard Index PFAS ranged from 3.9 to 52.4 percent of systems when broken down by state, with eight states exceeding 20 percent of systems. The percentage of systems that reported any PFAS ranged from 5.5 to 73.0 percent. Many systems and/or samples that were included in the aggregated state dataset did not monitor for all four Hazard Index PFAS. It is possible that more systems would have reported the presence of Hazard Index PFAS if they had monitored for all four Hazard Index PFAS. Additionally, as demonstrated in Table 10, when PFOA and/or PFOS were reported, at least one of the Hazard Index PFAS chemicals were also frequently reported. For systems that did not measure Hazard Index PFAS but measured and detected PFOA and/or PFOS, the groupwise analysis demonstrates that the Hazard Index PFAS were more likely to have been present in those systems as well. Table 11 presents system counts for systems where PFOS or PFOA were reported according to a) how many Hazard Index PFAS were monitored and b) how many Hazard Index PFAS were reported present.

Table 11: Non-Targeted State PFAS Finished Water Data – System Counts

According to Hazard Index PFAS Analyzed and Reported Present for Systems Where PFOS and PFOA were Reported

HI Analyzed	HI Reported Present					Total
	0	1	2	3	4	
1	148 (65.5%)	78 (34.5%)	-	-	-	226
2	138 (48.6%)	85 (29.9%)	61 (21.5%)	-	-	284
3	282 (36.5%)	183 (25.0%)	183 (25.0%	84 (11.5%)	-	732
4	511 (26.5%)	449 (23.3%)	668 (34.7%)	278 (14.4%)	20 (1.0%)	1,926
Total	1,079	795	912	362	20	

Among systems that reported the presence of PFOS and/or PFOA, the fraction of systems that also reported any Hazard Index PFAS tended to increase as systems monitored for more of the Hazard Index PFAS. At systems monitoring for a single Hazard Index PFAS, 34.5 percent reported a positive result at some point during sampling. This increased to 73.5 percent of systems reporting the presence of at least one Hazard Index PFAS when monitoring for all four Hazard Index PFAS. Not only did the fraction of systems reporting the presence of any Hazard Index PFAS increase as the number of Hazard Index PFAS monitored increased, so did the number of Hazard Index PFAS that were reported as present. When four Hazard Index PFAS were monitored, nearly 50 percent of systems reported the presence of two to three of the Hazard Index PFAS. Thus, if PFOS or PFOA are reported, there is a reasonable likelihood that multiple Hazard Index PFAS would be present as well.

b. Pairwise Chemical Co-Occurrence

In addition to considering the co-occurrence of six PFAS as two groups, the EPA conducted a pairwise analysis to further explore co-occurrence relationships. Table 12 shows the calculated system-level odds ratios for every unique pair of PFAS chemicals evaluated. The equation for calculating odds ratios is symmetrical. Because of this, in a given row it does not matter which chemical is ''Chemical A'' and which is ''Chemical B.'' Additional information on odds ratios may be found in USEPA (2024b) and a brief explanation is described following Table 12 as well as in section III.C of this preamble.

Table 12: Non-Targeted State PFAS Finished Water Data – System-level Counts of Pairwise Chemical Occurrence and Odds Ratios Calculated from Aggregated State Dataset PFAS Samples for PFOA, PFOS, and HI PFAS

Chem A	Chem B	Chems A and B Reported	Only Chem B Reported	Only Chem A Reported	Neither Chem Reported	Odds Ratio [95% CI]
HFPO-DA	PFBS	33	1,532	21	7,614	7.8 [4.5-13.5]
HFPO-DA	PFHxS	23	1,137	31	8,007	5.2 [3.1-8.9]
HFPO-DA	PFNA	20	327	34	8,818	15.9 [9.1-27.7]
HFPO-DA	PFOA	39	1,665	16	7,480	11.0 [6.2-19.5]
HFPO-DA	PFOS	37	1,530	18	7,613	10.2 [5.9-17.9]
PFBS	PFHxS	1,282	245	721	9,093	66.0 [56.4-77.2]
PFBS	PFNA	423	85	1,510	8,735	28.8 [22.7-36.6]
PFBS	PFOA	1,605	852	401	8,485	39.9 [35.0-45.4]
PFBS	PFOS	1,497	692	509	8,645	36.7 [32.4-41.7]
PFHxS	PFNA	415	108	1,115	9,455	32.6 [26.1-40.7]
PFHxS	PFOA	1,374	1,259	230	8,820	41.9 [35.9-48.7]
PFHxS	PFOS	1,369	939	235	9,140	56.7 [48.6-66.2]
PFNA	PFOA	575	2,190	23	8,764	100.1 [65.9-151.8]
PFNA	PFOS	555	1,864	43	9,089	62.9 [46.0-86.1]
PFOA	PFOS	2,304	341	729	9,972	92.4 [80.6-106.0]

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Odds ratios reflect the change in the odds of finding one chemical (*e.g.,* Chemical A) given that the second chemical (*e.g.,* Chemical B) is known to be present compared to the odds of finding it if the second chemical is not present. For example, as shown in Table 12, the point estimate of 92.4 for the odds ratio between PFOA and PFOS indicates that the odds of finding PFOA after knowing that PFOS has been observed are 92.4 times what the odds would have been if PFOS was not observed, and vice versa. For every pair of chemicals, both the point estimate and 95 percent confidence interval (CI) were above 1, indicating significant increases in the likelihood of detecting one chemical if the other is present.

Both as a group and as individual chemicals, the Hazard Index PFAS had a higher likelihood of being reported if PFOS or PFOA were present. PFHxS, PFNA, HFPO–DA, and PFBS (the individual Hazard Index PFAS) are demonstrated to generally co-occur with each other, as well. These data support that there is a substantial likelihood that PFHxS, PFNA, HFPO–DA, and PFBS co-occur in mixtures with a frequency of public health concern in drinking water systems as discussed in section III.C of this preamble.

D. Occurrence Relative to the Hazard Index

1. Proposal

In the proposed rule, the EPA analyzed the available state data in comparison to the proposed Hazard Index MCL of 1.0 to evaluate the co-occurrence of PFHxS, PFNA, HFPO–DA, and PFBS. The EPA requested comment on the number of systems estimated to solely exceed the Hazard Index (but not the PFOA or PFOS MCLs) according to the approach outlined in USEPA (2024b).

2. Summary of Major Public Comments and EPA Responses

The EPA received comments on the analyses presented in the proposal of occurrence relative to the Hazard Index. Many commenters agreed that the Hazard Index PFAS co-occurred in mixtures at levels of health concern. Two of these comments came from states that conducted monitoring of Hazard Index PFAS post-UCMR 3 and stated that those occurrence data supported the EPA's findings. Several state agencies provided a summarized analysis of the number of systems expected to exceed the proposed Hazard Index of 1.0 in their state. The EPA notes that these estimates were based on the proposed Hazard Index, which included two significant figures. Since the EPA has determined to finalize the Hazard Index with one significant figure, these estimations are likely high. Nonetheless, these state data and the analyses provided by commenters provide illustrative confirmatory insight of the EPA's Hazard Index analyses (please see section IV of this preamble for additional discussion on the usage of significant figures).

One commenter suggested that a national dataset and model complete with all four Hazard Index PFAS are necessary to accurately estimate the number of systems that may exceed the Hazard Index. The EPA disagrees with the commenter; as described in section F, state data and model outputs were appropriately combined to estimate exceedance of the Hazard Index on a national level. Several commenters stated that there was a limited amount of available data to determine the prevalence of co-exposure of the Hazard Index compounds, and that further review would be needed prior to establishing the Hazard Index. The EPA disagrees with these commenters and believes that sufficient data were available to reasonably assess the occurrence of Hazard Index PFAS. An analysis of co-occurrence of Hazard Index compounds using a substantial amount of data encompassing tens of thousands of samples across over 10,000 systems is provided in section VI.C. of this preamble above and demonstrates that the four Hazard Index PFAS co-occur with each other as well as with PFOA and PFOS. One commenter suggested that more systems may exceed the Hazard Index than the PFOA and PFOS MCLs, since current treatment technologies have been optimized for PFOA and PFOS and not for other PFAS. The EPA's analysis of state datasets clearly contradicts this claim; using the best available data and scientifically robust analytical approaches, the EPA estimates more systems will exceed the PFOA and PFOS MCLs than the Hazard Index MCL. The use of a single significant figure for the Hazard Index MCL in this final rule will further increase the likelihood of this being the case.

3. Final Rule

The EPA used its updated state dataset to update analyses related to Hazard Index occurrence and found the analyses generally consistent with the proposal analyses. In the final rule, the EPA is reducing the number of significant figures used to determine Hazard Index exceedance following all calculations and rounding from two to one; this change had the effect of reducing system counts expected to exceed the Hazard Index. For purposes of the final analyses, only systems with an unrounded Hazard Index of 1.5 or greater were counted as an exceedance. Table 13 presents the total number and percentage of monitored systems with results above the proposed Hazard Index MCL based on state reported Hazard Index PFAS data for the states that conducted non-targeted monitoring and that sampled all four Hazard Index PFAS as a part of their overall monitoring efforts. The EPA notes that for equivalent comparison purposes Table 13 only accounts for samples that included reported values (including non-detects) of all four Hazard Index PFAS. As shown within the table, the majority of states evaluated had monitored systems with results above the proposed Hazard Index MCL, ranging from 0.35 to 3.17 percent of total monitored systems. For additional discussion on the usage of significant figures in this rule, please see section IV of this preamble.

Table 13: Non-Targeted State PFAS Finished Water Data – Summary of Total Number and Percent of Monitored Systems Exceeding the Hazard Index with Samples Containing Reported Values of All Four Hazard Index PFAS

State	Total Monitored Systems > Final HI of 1	Percent Systems > Final HI of 1
Colorado	2	0.50%
Illinois	7	0.70%
Indiana	0	0.00%
Kentucky	2	2.70%
Maryland	2	3.17%
Massachusetts	23	1.76%
Michigan	17	0.68%
Missouri	1	0.91%
New York	7	1.28%
New Hampshire	3	2.17%
North Dakota	0	0.00%
Ohio	16	1.08%
South Carolina	2	0.68%
Vermont	2	0.35%
Wisconsin	7	3.03%

Further evaluating the available state data related to the proposed Hazard Index MCL of 1, Table 14 presents the total number of systems that exceed the final Hazard Index of 1 based on state reported Hazard Index PFAS results for the same states shown in Table 13. However, in this case, the EPA also analyzed the same non-targeted state data, including additional samples even if those samples did not contain reported values (including non-detects) for all four Hazard Index PFAS (*i.e.,* exceeding the Hazard Index based on two or three Hazard Index PFAS with reported values included within a sample). Moreover, while these states did monitor for all four Hazard Index PFAS as a part of their overall monitoring, in a subset of those states some samples did not include reported data on all four Hazard Index PFAS (*i.e.,* values of one or more of the Hazard Index PFAS were not reported as non-detect, rather no value was reported). This analysis, presented in Table 14, shows an increase in the number of monitored systems exceeding the proposed Hazard Index of 1 and demonstrates prevalence of these PFAS at levels of concern, even when all four PFAS may not be included within a sample.

Table 14: Non-Targeted State PFAS Finished Water Data – Summary of Total Monitored Systems Exceeding the Hazard Index with Samples Containing Reported Values of 2 or More Hazard Index PFAS

State	Total Monitored Systems > Final HI of 1	Percent Systems > Final HI of 1
Colorado	2	0.50%
Illinois	7	0.70%
Indiana	0	0.00%
Kentucky	2	2.70%
Maine	4	0.62%
Maryland	7	5.19%
Massachusetts	31	2.34%
Michigan	17	0.68%
Missouri	1	0.87%
New Jersey	27	4.06%
New York	18	2.67%
New Hampshire	17	3.04%
North Dakota	0	0.00%
Ohio	16	1.08%
South Carolina	2	0.67%
Vermont	2	0.35%
Wisconsin	7	2.95%

Combining the non-targeted monitoring results shown previously with targeted state monitoring conducted for all four Hazard Index PFAS showed at least 864 samples from 211 PWSs in 21 states had results above the final Hazard Index of 1. These systems serve approximately 4.7 million people. More information on occurrence in state monitoring is available in section III.C of this preamble and in USEPA (2024b).

In summary, the finished water data collected under both non-targeted and targeted state monitoring efforts from 32 states showed there are at least 1,772 PWSs serving a total population of approximately 24.3 million people that have at least one result exceeding the final PFOA MCL of 4.0 ng/L. In those same 32 states, there are also at least 1,432 PWSs serving a total population of approximately 21.0 million people that have at least one result exceeding the final PFOS MCL of 4.0 ng/L. Finished water data showed that there are at least 187 systems in 23 states serving a total population of approximately 4.4 million people with at least one result exceeding the final PFHxS MCL of 10 ng/L. Finished water data from 12 states showed there are at least 52 systems serving a total population of approximately 176,000 people that have at least one result exceeding the final PFNA MCL of 10 ng/L. Finished water data showed 13 systems from 5 states serving over 226,000 people have at least one result exceeding the final HFPO–DA MCL of 10 ng/L. Related to the Hazard Index, finished water data collected under both non-targeted and targeted state monitoring efforts in 21 states showed there are at least 211 systems serving a total population of approximately 4.7 million people with results above the final Hazard Index value of 1 for PFHxS, PFNA, HFPO–DA, and PFBS. Samples that only had monitoring results for one Hazard Index PFAS were not included. USEPA (2024b) presents a detailed discussion on state PFAS monitoring information.

E. Occurrence Model

A Bayesian hierarchical occurrence model was developed to characterize national occurrence of the four PFAS that were most frequently detected in the UCMR 3: PFOA, PFOS, PFHxS, and PFHpA.[8] This model was used to generate the baseline national occurrence estimates for PFOA, PFOS, and PFHxS, which were used in the subsequent economic analysis in USEPA (2024g). Bayesian hierarchical models are a widely used statistical approach in which subsets of data may be recognized as more related than others (such as samples from the same PWS are more related than samples between different PWSs) to capture complex relationships between levels of data and can aid in understanding the factors that influence outcomes. The objective of this model was to use both UCMR 3 data and supplemental state data to develop national estimates of

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[8] PFHpA was included in the model because of its UCMR 3 occurrence data availability.

PFAS occurrence that inform occurrence distributions both within and across PWSs. Supplemental state data were incorporated to improve the model's ability to estimate PFAS occurrence at levels below the UCMR 3 minimum reporting levels (20 ng/L for PFOA, 40 ng/L for PFOS, and 30 ng/L for PFHxS). The state data incorporated to supplement the model came from publicly available datasets. In order to maintain the statistically robust UCMR 3 sampling framework, thereby enabling the agency to make conclusions about national representativeness of the model results, incorporation of state data into the model was limited only to data from systems that took part in the UCMR 3. The model does not include PFNA and PFBS due to data limitations; PFNA and PFBS lacked sufficient reported values above the UCMR 3 minimum reporting levels to be incorporated into the model. The model has been peer reviewed and is described extensively in Cadwallader et al. (2022).

The model uses Markov chain Monte Carlo (MCMC) simulation and the assumption of lognormality in PFAS chemical occurrence. Markov chain Monte Carlo is a powerful statistical tool used to understand uncertainty and making informed decisions when analyzing data. The EPA has used similar hierarchical models to inform regulatory decision making in the past, such as for development of the NPDWR for Arsenic and *Cryptosporidium parvum* (USEPA, 2006c; USEPA, 2000e).

After log-transformation of data informing the model, system-level means (where each system has a mean concentration for each chemical) were assumed to be distributed multivariate normally. Further, within-system occurrence was assumed to be distributed normally for each chemical. Since system-level means were modeled multivariate normally, correlation between estimated system-level means across chemicals could also be assessed. The assumption of lognormality as well as the incorporation of state data with lower reporting limits allowed the model to generate reasonable estimates for PFAS occurrence at levels below the UCMR 3 minimum reporting levels.

After the model was fit with available data from PWSs that were included in the UCMR 3, it was used to simulate occurrence at an inventory of active community water systems (CWS) and non-transient non-community water systems (NTNCWS) extracted from the Safe Drinking Water Information System (SDWIS). System-level means for non-UCMR 3 systems were simulated by sampling from the multivariate normal distribution of system-level means that was produced during the model fitting process. For systems that were included in the UCMR 3, the fitted system-level mean was used directly. This approach allowed national occurrence distributions to be estimated alongside the associated populations when combined with population data from SDWIS.

1. Proposal

In the March 2023 proposal preamble, model estimates of contaminant occurrence were presented. For the analysis presented in the proposal, UCMR 3 data were supplemented with 23,130 analytical results from 771 systems across 17 states that were available from public state websites through August 2021. Key model results that were presented directly included correlation coefficients across pairs of chemicals included in the model, extrapolated estimates of the number of system level means anticipated to exceed various threshold, and the estimated population associated with systems that had mean concentrations exceeding the various thresholds. The results indicated that system-level mean concentrations were moderately to strongly correlated across the modeled PFAS and that thousands of systems were estimated to have mean PFAS concentrations in the range of single digit ng/L.

2. Summary of Major Public Comments and EPA Responses

A few commenters stated that they believed the model was an overly complicated approach to characterizing chemical occurrence and found it difficult to understand. Further, a few commenters stated that they believed the model was not transparent. The EPA disagrees; the occurrence approach used by the agency in this rule is based on a widely utilized and accepted statistical approach which is used in a variety of fields from education to health care and from business to the environment. These models allow exploration of the relationships among groups of data and the EPA used this model to better inform the agency's understanding of probable PFAS occurrence. For more information about Bayesian statistics and the wide variety of potential applications, see, for example, Hoff (2009); van de Schoot et al. (2021); Aguilera et al. (2011); and Messner et al. (2001). While the model uses an advanced statistical method and requires some statistical background to fully understand, Bayesian hierarchical models have previously been employed to assess occurrence for drinking water contaminants, as was discussed in the March 2023 proposal preamble as well as Cadwallader et al. (2022). Cadwallader et al. (2022) describes the model structure while the annotated model code and inputs were provided directly as supporting information alongside the manuscript. This information was incorporated into the docket for this rule's proposal. Sufficient information to replicate the model run was provided. Thus, the agency disagrees with the assertion that the model was not transparent.

Regarding the model complexity, the core structure of this specific model is comparatively simple among Bayesian hierarchical models. The model uses a multivariate normal distribution of system-level means (of log transformed data) for the four modeled PFAS. It also includes a parameter for small systems to assess whether they appear to have systematically different (higher or lower) concentrations than large systems. As stated in Cadwallader et al. (2022), the model extrapolates to the nation by sampling from the multivariate normal distribution and accounting for whether the system being simulated was small. The multivariate normal distribution and the parameter to distinguish small systems from large systems are two simple but important pieces of the model structure.

Many commenters stated that the model relied on insufficient data and produced substantial underestimates of the number of systems that would fail to meet MCL requirements. The agency disagrees both that the approach taken would systematically underestimate PFAS occurrence and that the data were insufficient inform the model. The Bayesian approach used here makes a precedented assumption about drinking water contaminant occurrence distributions (lognormality) and uses the available data to generate iterative estimates of distribution parameters that capture uncertainty through MCMC simulation. Across these iterations, the density of the posterior distribution for model parameters is proportionate to the likelihood that a given value would have produced the observed data. The subsequent national extrapolations also reflect this uncertainty.

For the results presented in the March 2023 proposal preamble, the model was fit using 171,017 analytical results across the 4,920 UCMR 3 systems. This was a nationally representative set of systems. 147,887 of the analytical results were collected as part of UCMR 3 while 23,130 were aggregated from 17 subsequently collected state datasets. The model was designed to utilize both results reported as observed concentrations (8,209 results) and results reported as less than a reporting limit (162,808 results). While the UCMR 3 used higher reporting limits than are currently available, both reported concentrations and values reported as below the minimum reporting level cumulatively make substantial contributions to informing the model's estimates of the PFAS occurrence distribution because of this statistically robust framework. Due to this efficient use of data, and the steps taken to maintain a nationally representative set of systems, the agency believes that the over 170,000 analytical results were sufficient to generate reasonable estimates of occurrence for the modeled contaminants.

Several commenters expressed concern with model bias resulting from the supplemental state data that was incorporated when fitting the model. The hierarchical structure of the model minimizes the bias impact of introducing additional state data for only some UCMR 3 systems (those with additional data available) because the data are explicitly linked to their parent systems rather than being pooled with all other data informing the model. The primary impact that these data have is on the model's estimate of specific system means for those systems that had additional data and informing the within-system variability parameters in the model. Refinement of a single system's mean estimate has a much smaller impact on the high-level distribution of system-level means and such shifts are proportionate to the added evidence derived from the supplemental data.

The addition of data from systems not included in the UCMR 3 would pose a much greater concern for bias, since not all states have publicly available data. States with additional data would become disproportionately represented in the fit of the high-level distribution, since each system acts as a data point in fitting the distribution. The resulting high-level distribution would shift to resemble the states more closely with higher system representation in the source dataset. This would also be reflected in the subsequent national extrapolation. This same bias concern applies to national extrapolation approaches where some fraction of systems in a subset are identified as exceeding a given threshold and the national inventory of systems is multiplied by that fraction to generate a national estimate of systems that would exceed the threshold. If certain states have a disproportionate number of systems included in the subset compared to in the nation as a whole, the national estimate will be biased towards the tendencies of those states. In addition to this bias, the simple example approach discussed above would not naturally reflect uncertainty. Thus, for the purpose of national extrapolation, a nationally representative set of systems is more appropriate, even if data from other systems are available.

While the EPA believes the model design and data selected for the analysis presented in the March 2023 proposal remain appropriate given the data availability at the time, the EPA has also continued to collect newly available data from publicly available state datasets, as the agency committed to in the proposed rulemaking (USEPA, 2023f). The Bayesian hierarchical model has been refit using the updated dataset with the same methods and criteria for data selection that were used for the analysis presented in the March 2023 proposal.

3. Final Rule

After considering public comment, the agency has used the Bayesian statistical model described in Cadwallader et al. (2022) to support the economic analysis for this final regulation by combining the available occurrence information from UCMR 3 and state data subsequently collected at UCMR 3 systems to maintain the nationally representative nature of the set of drinking water systems informing the model, utilizing those data to compute estimates of national occurrence for PFAS contaminants, and providing estimates on the number of systems impacted by this final rule. These estimates directly informed the economic analysis in USEPA (2024g). For the final rule, the model was updated with additional state data collected through May 2023. In total, based on public comment, the EPA supplemented the state dataset with 65,537 analytical results from 1,156 systems across 28 states. Of these supplemental data, 24,950 analytical results were observed concentrations while 40,587 results were reported as below some reporting limit. The previously presented results have been updated and are presented in Table 15. The EPA notes that results from the updated dataset and model were confirmatory of its proposal analyses and did not result in changes to the EPA's final decisions. Median estimates and 90 percent credible intervals are shown for counts of systems with system-level means at or above various PFAS concentrations in Table 15 and the population served by those systems in Table 16.

Table 15: National Occurrence Model Estimate – Estimated Number of Systems

With System-level Means at or Above Various Concentrations

Concentration (ng/L)	**PFHxS [90% CI]**	**PFOA [90% CI]**	**PFOS [90% CI]**
4.0	1,828 [1,226-2,689]	3,260 [2,416-4,349]	3,368 [2,461-4,566]
5.0	1,252 [823-1,888]	2,194 [1,588-2,994]	2,447 [1,757-3,386]
10.0	340 [209-555]	523 [354-771]	793 [537-1,166]

Table 16: National Occurrence Model Estimate – Estimated Population Served by Systems with System-level Means at or Above Various Concentrations

Concentration (ng/L)	PFHxS [90% CI]	PFOA [90% CI]	PFOS [90% CI]
4.0	20,386,000 [17,436,000-24,351,000]	34,343,000 [30,897,000-40,600,000]	34,313,000 [30,703,000-41,110,000]
5.0	15,436,000 [12,524,000-18,458,000]	24,287,000 [21,551,000-28,222,000]	26,594,000 [23,793,000-31,240,000]
10.0	4,645,000 [3,557,000-7,205,000]	7,132,000 [4,871,000-8,987,000]	10,205,000 [7,552,000-12,232,000]

For PFOA, PFOS, and PFHxS, thousands of systems were estimated to have mean concentrations over the lowest thresholds (*i.e.,* 4.0 and 5.0 ng/L) presented in Tables 15 and 16 with the total population served estimated to be in the tens of millions. The populations shown here represent the entire populations served by systems estimated to have system-level means over the various thresholds. It is likely that different subpopulations would be exposed to different mean PFAS concentrations if multiple source waters are used.

In addition to the estimates of individual chemical occurrence, the multivariate normal distribution of system-level means allowed the model to provide insight on estimated co-occurrence. The model results support the co-occurrence of PFOA, PFOS and Hazard Index PFAS. The model evaluated whether untransformed (*i.e.,* expressed in the original units of measurement) estimates of system-level means were correlated across each unique pair of the four modeled chemicals included in the model. Estimates of the Pearson correlation coefficient are shown in Table 17. The Pearson correlation coefficient serves as an indicator of the strength of the linear relationship between two variables and may range from −1 to 1. Positive values indicate a positive relationship (*i.e.,* as one variable increases, so does the other). shown in Table 17. The Pearson correlation coefficient serves as an indicator of the strength of the linear relationship between two variables and may range from −1 to 1. Positive values indicate a positive relationship (*i.e.,* as one variable increases, so does the other).

Table 17: National Occurrence Model Estimate – Median Estimated Pearson Correlation Coefficient and 90% Credible Interval Among System-level Means

Chemical Pair	Pearson Correlation Coefficient [90% CI]
PFOS-PFOA	0.73 [0.63-0.80]
PFOS-PFHpA	0.67 [0.56-0.75]
PFOS-PFHxS	0.82 [0.72-0.89]
PFOA-PFHpA	0.83 [0.79-0.87]
PFOA-PFHxS	0.51 [0.39-0.60]
PFHpA-PFHxS	0.58 [0.44-0.67]

The EPA considered a moderate strength correlation as greater than 0.5 and a strong correlation as greater than 0.7. Each point estimate of correlation coefficients between two chemicals was above the threshold for a moderate strength correlation. The carboxylic acids (PFOA–PFHpA) and sulfonic acids (PFOS–PFHxS) had the highest estimated correlation strengths, with both the point estimate and the 90 percent credible interval above the threshold for a strong correlation. PFOS–PFOA and PFOS–PFHpA had similar point estimates and 90 percent credible interval ranges, spanning the moderate-to-strong correlation range. Both PFOA–PFHxS and PFHpA–PFHxS had the bulk of their posterior distributions fall in the range of a moderate strength correlation. Thus, the model predicted significant positive relationships among system-level means of all four chemicals that were included. These results support the co-occurrence discussion presented in section VI.C of this preamble that indicated extensive co-occurrence of PFOA, PFOS, and the Hazard Index PFAS observed in state datasets from both groupwise and pairwise chemical perspectives.

F. Combining State Data With Model Output To Estimate National Exceedance of Either MCLs or Hazard Index

In order to broadly estimate the number of systems that would be impacted by the regulation, including MCLs of 4.0 ng/L for PFOA and PFOS alongside a Hazard Index of 1 for PFHxS, PFNA, HFPO–DA, and PFBS, findings from non-targeted monitoring in state datasets were combined with model estimates. Specific details on the methodology can be found in USEPA (2024b). Briefly, information collected from non-targeted state datasets included the fractions of systems that reported a measurement at or above the UCMR 5 minimum reporting level for a given analyte and an empirical cumulative distribution function (eCDF) consisting of system-level maximum observed concentrations of that chemical at these systems. The UCMR 5 minimum reporting levels for PFNA, HFPO–DA, and PFBS are equivalent to 4 ng/L, 5 ng/L, and 3 ng/L, respectively (USEPA, 2022j). This applies the assumption that the fraction of systems that observed PFNA, HFPO–DA, and PFBS at or above UCMR 5 minimum reporting levels and the maximum concentrations observed at those systems are reasonably representative of the nation.

1. Proposal

The model was used to simulate EP-level concentrations of the four modeled PFAS (PFOA, PFOS, PFHpA, and PFHxS) under the assumption that within-system concentrations are lognormally distributed (a common assumption for drinking water contaminants, see (Cadwallader et al. (2022)) and that variability in concentrations is entirely across EP (thus a given EP is assumed to have a constant concentration). For each system, the maximum estimated EP PFOA or PFOS concentration was selected to determine whether the system exceeded either of the proposed MCLs of 4.0 ng/L. The EP with the maximum concentration is the point that determines whether a system has an EP that is above an MCL. Estimates of the system-level maximum for PFHxS were also selected for the Hazard Index calculation. The maximum value of the sum of the four modeled PFAS at each system was selected and used as a basis for determining which systems would receive superimposed concentrations of the three remaining Hazard Index chemicals (PFNA, HFPO–DA, and PFBS). This approach was selected due to the extensive observed co-occurrence of PFAS in the UCMR 3, state data, and modeled estimates.

Multiple methods of system selection were used that reflected different degrees of co-occurrence. The chemical concentration that was applied to selected systems were randomly sampled from the eCDF for each chemical. Based on the model output, this assumes that system-level maximums for PFNA, HFPO–DA, and PFBS would occur at the same location within a system. Given the substantial co-occurrence among PFAS observed and estimated across various analyses, combination of system-level maximums independently pulled from chemical eCDFs is a reasonable simplifying assumption. This is particularly true since systems selected for each chemical are not necessarily the same and in most cases were probability weighted. Estimates of the range of systems impacted were developed by taking Q5 and Q95 estimates for each method. The low end of the range was taken as the lowest Q5 estimate across methods, rounded down, while the high end of the range was taken as the highest Q95 estimate across methods, rounded up. This was also done for the total population served by these systems.

The analysis to support the March 2023 proposal estimated that 100–500 systems that were not already exceeding an MCL for PFOA or PFOS would exceed the Hazard Index. This resulted in a total of 3,400–6,300 systems estimated to be exceeding either the Hazard Index, the MCL for PFOA, or the MCL for PFOS.

2. Summary of Major Public Comments and EPA Responses

One commenter stated that they believed it is difficult to determine whether the estimated number of systems exceeding the Hazard Index is a reasonable estimate until a complete national dataset is available. The EPA disagrees with this commenter. The agency believes that it has taken steps to produce reasonable estimates using a robust set of available data, and that the data and analyses are sufficient to inform the EPA's regulatory decisions. Namely, this includes the use of non-targeted state datasets and multiple scenarios reflecting varying degrees of co-occurrence as described in USEPA (2024b). Among other important uses for these data, the EPA considered them to inform the regulatory determination for the mixture of the Hazard Index PFAS and the EA. The EPA has used these data to clearly demonstrate that there is a substantial likelihood that combinations of the Hazard Index PFAS co-occur as mixtures in public water systems with a frequency and at levels of public health concern. See section III of this preamble for additional discussion. Additionally, these data support the EPA's EA, and considerations of costs and benefits consistent with SDWA's requirements. See section XII of this preamble for further discussion.

3. Final Rule

The method to combine state data for non-modeled Hazard Index PFAS with model estimates has largely remained the same for this final rule as it was for the March 2023 proposal. One key change, based on public comments, was to use an updated set of non-targeted state data to inform Hazard Index contaminant prevalence above UCMR 5 minimum reporting levels and eCDFs. Another key alteration, also based on public comments, was accounting for significant figures when counting systems exceeding the MCL for PFOA, the MCL for PFOS or the Hazard Index. For a system to be exceeding the Hazard Index, it must be greater than or equal to 2 (*i.e.,* greater than 1) after rounding (for additional discussion on significant figure usage in the final rule, please see section IV of this preamble). To exceed the MCLs for PFOA or PFOS, the concentration must be greater than or equal to 4.1 ng/L after rounding. Finally, model estimates of PFHxS were converted to zero for the purposes of calculating the Hazard Index if they fell below the PQL of 3 ng/L.

The total number of systems estimated to be exceeding one or more MCLs in the rule was 4,100–6,700 (compared to 3,400–6,300 in the proposal) serving a total population of 83–105 million people. Among these systems, 100–300 are estimated to be exceeding the Hazard Index without exceeding the PFOA or PFOS MCLs. The EPA used these modeled estimates to inform the costs and benefits determination as described in section XII of this preamble. Additional details regarding the approach used here can be found in USEPA (2024b).

G. UCMR 5 Partial Dataset Analysis

1. Summary of Major Public Comments and EPA Responses

UCMR 5 occurrence data were not available to inform the proposal, but the agency discussed that additional nationwide monitoring data would be available for systems participating in the monitoring program. Some commenters called for the EPA to delay issuance of the final PFAS rule until the complete UCMR 5 occurrence dataset can be analyzed, and some commenters stated that rule promulgation should be delayed until at least a portion of the UCMR 5 data is obtained. The EPA disagrees with these commenters. The EPA is not required under the statute to wait for another round of UCMR data to be collected before proposing or finalizing a regulation; in this case, the completion of UCMR 5 data reporting is expected at the end of 2025, with the final dataset not being available until 2026. Rather, SDWA section 1412(b)(1)(B)(ii)(II) expressly provides that the EPA must use the ''best *available* public health information'' in making a regulatory determination (emphasis added). The EPA has sufficiently robust occurrence information to make regulatory determinations and promulgate a regulation for the six PFAS in this regulation. In addition to serving as a significant way for helping many utilities reduce initial monitoring costs, the final full UCMR 5 dataset will also be valuable for informing future regulatory decisions for the 23 PFAS included in UCMR 5 that are not directly addressed by this rulemaking. The agency believes that the best currently available occurrence data demonstrate sufficient occurrence or substantial likelihood of occurrence for the contaminants included in the final rule.

2. Final Rule

While the EPA is under no legal obligation to consider the preliminary, partial UCMR 5 dataset prior to rule promulgation, based on public comment and interest, the agency examined UCMR 5 data released as of February 2024 (USEPA, 2024n). While these data were not available for this rule's proposal, are not complete, and are not a basis for informing the agency's decisions for the final rule, the EPA notes that they generally confirm the extensive occurrence analyses the agency has conducted: namely, that all six regulated PFAS occur in finished drinking water and that the six regulated PFAS co-occur with one another. The EPA notes some important caveats when considering these data. First, as of February 2024, the partial UCMR 5 dataset is a subset of data that will be collected, representing approximately 24 percent of the total data that might be collected under that effort. Additionally, under UCMR 5, systems must collect either 2 or 4 samples, depending on their source water characteristics. In this preliminary dataset, systems have varying degrees of completeness in their sample collection and results may shift at the system level as additional samples are collected. Analyses included examination of sample-level results as well as EP mean-level results.

The UCMR 5 data publicly available as of February 2024 included a combined total of 100,629 analytical results for PFOA, PFOS, PFHxS, PFNA, HFPO–DA, and PFBS ranging from 16,766 to 16,778 analytical results for each chemical. 16,743 complete sample sets where an analytical result was reported for each chemical were available. 9,528 EPs and 3,719 PWS had at least one analytical result for each of the six PFAS and one sample for which the Hazard Index could be calculated. As mentioned previously, this partial dataset is estimated to contain approximately 24 percent of the data that will be available once the dataset is completed and finalized.

The preliminary dataset was assessed for sample-level threshold exceedances of PFOA (4.0 ng/L), PFOS (4.0 ng/L), PFHxS (10 ng/L), PFNA (10 ng/L), HFPO–DA (10 ng/L), and the Hazard Index (1). Note that for PFOA and PFOS, two significant figures were considered (*i.e.,* analytical results had to meet or exceed 4.05 to be considered exceedances) while for PFHxS, PFNA, HFPO–DA, and the Hazard Index one significant figure was considered (*i.e.,* an analytical result had to meet or exceed 15 to be considered an exceedance for PFHxS, PFNA, and HFPO–DA and 1.5 to be considered an exceedance for the Hazard Index). Sample-level analysis only included complete sample sets while EP and system-level analysis included only systems that provided sufficient data to determine maximum PFOA, PFOS, PFHxS, PFNA, and HFPO–DA, and Hazard Index (which required at least one sample set where the Hazard Index could be calculated). The EPA notes that this analysis does not represent an estimate for the number of systems that will be in compliance with the MCL; as discussed in section V of this preamble, MCL compliance is determined based on an RAA. Additionally, samples below the PQL would be treated as zero in the compliance calculation. In the preliminary UCMR 5 dataset, PFOA exceeded 4.0 ng/L in 6.1 percent of samples (1,024 samples), at 7.5 percent of EPs (719 EPs), and at 11.2 percent of systems (415 systems). PFOS exceeded 4.0 ng/L in 6.6 percent of samples (1,100 samples), at 8.0 percent of EPs (766 EPs), and at 12.4 percent of systems (462 systems). PFHxS exceeded 10 ng/L in 0.4 percent of samples (66 samples), at 0.6 percent of EPs (53 EPs), and at 1.1 percent of systems (42 systems). PFNA exceeded 10 ng/L in <0.1 percent of samples (5 samples), at <0.1 percent of EPs (5 EPs), and at 0.1 percent of systems (5 systems). HFPO–DA exceeded 10 ng/L in <0.1 percent of samples (2 samples), at <0.1 percent of EPs (1 EP), and at <0.1 percent of systems (1 system). The Hazard Index exceeded 1 in 0.5 percent of samples (76 samples), at 0.6 percent of EPs (60 EPs), and at 1.3 percent of systems (48 systems). When the thresholds were considered simultaneously, 9.0 percent of samples (1,504 samples), 10.9 percent of EPs (1,043 EPs), and 15.8 percent of systems (589 systems) exceeded a threshold. Note that single sample exceedances of thresholds do not necessarily reflect the averages that might be observed in the completed dataset. Specifically, the EPA notes that it is likely that many of the 15.8 percent of systems with an exceedance would not exceed the MCLs because additional samples used to determine an RAA may produce lower results.

To further illustrate this point, though there is insufficient data to fully evaluate RAAs,[9] EP-level means and systems with EP-level means exceeding an MCL threshold were also assessed with the preliminary dataset. For this analysis, only complete sample sets and EPs with multiple complete sample sets were included. 5,269 EPs and 2,498 systems had data that met these criteria. When calculating EP means, results reported as less than the minimum reporting limit were treated as zero. Note that for PFOA and PFOS, two significant figures were considered (*i.e.,* calculated means had to meet or exceed 4.05 to be considered exceedances) while for PFHxS, PFNA, HFPO–DA, and the Hazard Index one significant figure was considered (*i.e.,* calculated mean had to meet or exceed 15 to be considered an exceedance for PFHxS, PFNA, and HFPO–DA and 1.5 to be considered an exceedance for the Hazard Index). Mean PFOA concentration exceeded 4.0 ng/L at 4.8

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[9] An RAA is calculated using results for samples taken at a particular monitoring location during the previous four consecutive quarters (see section XIII.B for more information).

percent of EPs (253 EPs) and at 6.0 percent of systems (149 systems). Mean PFOS concentration exceeded 4.0 ng/L at 5.3 percent of EPs (278 EPs) and at 7.2 percent of systems (179 systems). Mean PFHxS concentration exceeded 10 ng/L at 0.3 percent of EPs (15 EPs) and at 0.4 percent of systems (11 systems). Mean PFNA concentration exceeded 10 ng/L at <0.1 percent of EPs (1 EP) and at <0.1 percent of systems (1 system). Mean HFPO–DA concentration exceeded 10 ng/L at <0.1 percent of EPs (1 EP) and at <0.1 percent of systems (1 system). Mean Hazard Index exceeded 1 at 0.3% of EPs (18 EPs) and at 0.6% of systems (14 systems). Considered simultaneously, an MCL was exceeded at 7.2 percent of EPs (381 EPs) and 9.4 percent of systems (235 systems). While the EP means described above include multiple sample sets, observed mean concentrations are likely to change as systems complete UCMR 5 sampling.

Among 16,743 completed sample sets and 9,529 EPs and 3,719 systems which had at least one result for each analyte, 13.9 percent of samples (2,335 samples), 16.5 percent of EPs, and 22.6 percent of systems (842 systems) had an observed concentration at or above the minimum reporting level for at least one of the 6 PFAS. Table 18 shows counts of samples, EPs, and systems according to how many of the 6 PFAS included in this final rule were present at or above the minimum reporting level. As shown in Table 18, about 7.5 percent of samples, 9.4 percent of EPs, and 14.2 percent of systems observed multiple PFAS at or above the minimum reporting level.

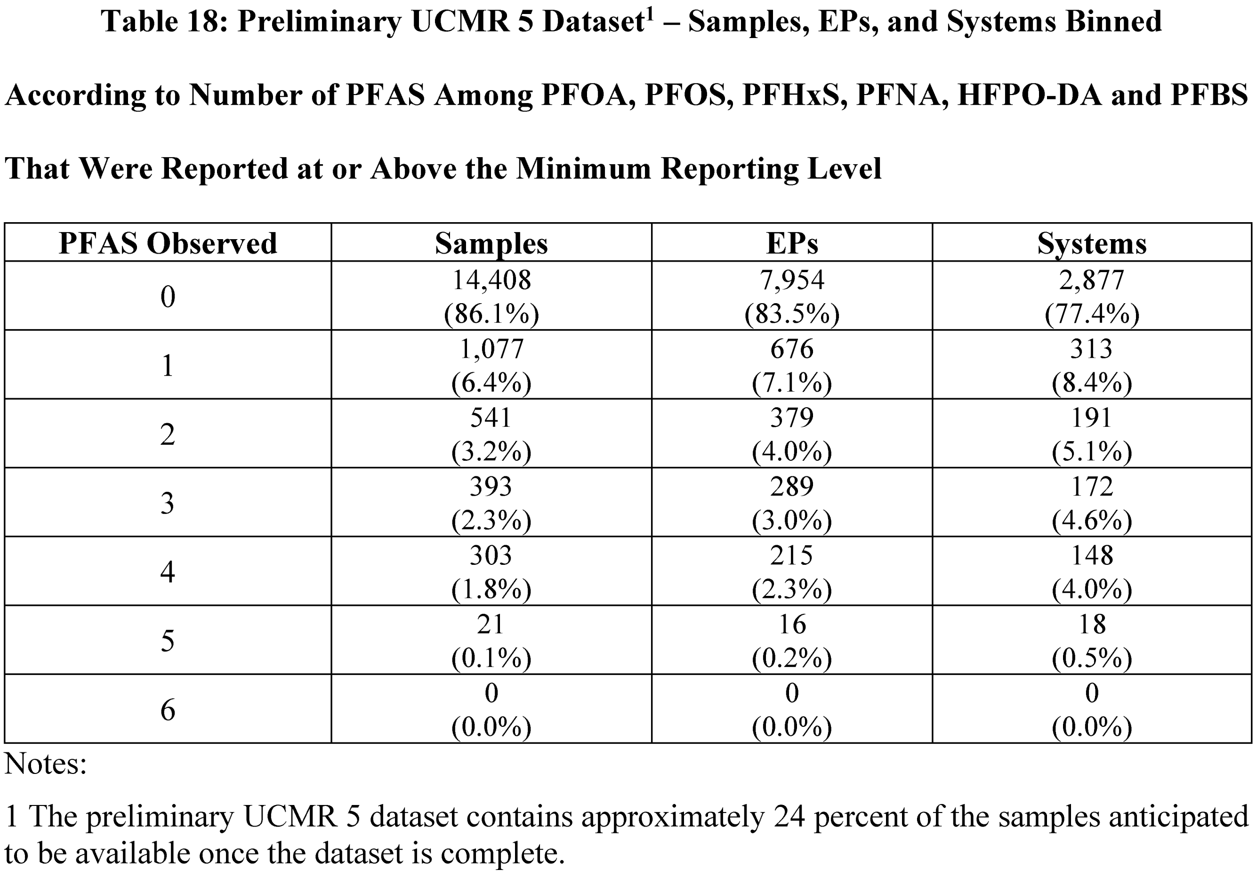

Table 18: Preliminary UCMR 5 Dataset[1] – Samples, EPs, and Systems Binned According to Number of PFAS Among PFOA, PFOS, PFHxS, PFNA, HFPO-DA and PFBS That Were Reported at or Above the Minimum Reporting Level

PFAS Observed	Samples	EPs	Systems
0	14,408 (86.1%)	7,954 (83.5%)	2,877 (77.4%)
1	1,077 (6.4%)	676 (7.1%)	313 (8.4%)
2	541 (3.2%)	379 (4.0%)	191 (5.1%)
3	393 (2.3%)	289 (3.0%)	172 (4.6%)
4	303 (1.8%)	215 (2.3%)	148 (4.0%)
5	21 (0.1%)	16 (0.2%)	18 (0.5%)
6	0 (0.0%)	0 (0.0%)	0 (0.0%)

Notes:

1 The preliminary UCMR 5 dataset contains approximately 24 percent of the samples anticipated to be available once the dataset is complete.

Groupwise co-occurrence was also examined in the preliminary UCMR 5 dataset. Table 19 provides the counts and percentages of systems, EPs, and samples where PFOA and/or PFOS were reported as well as whether any of the Hazard Index PFAS were reported. Sample-level results only included completed sample sets while system-level results only included systems which provided one analytical result for each of the 6 PFAS.

Table 19: Preliminary UCMR 5 Dataset[1] – Samples, EPs, and Systems Binned According to Whether PFOS or PFOA were Reported by States and Whether Additional Hazard Index PFAS were Reported

Type	No PFOS or PFOA Reported		PFOS or PFOA Reported		Total Count
	No HI PFAS Reported	At Least One HI PFAS Reported	No HI PFAS Reported	At Least One HI PFAS Reported	
Samples	14,408 (86.1%)	786 (4.7%)	498 (3.0%)	1,051 (6.3%)	16,743
EPs	7,954 (83.5%)	508 (5.3%)	317 (3.3%)	750 (7.9%)	9,529
Systems	2,877 (77.4%)	242 (6.5%)	145 (3.9%)	455 (12.2%)	3,719

Notes:

[1] The preliminary UCMR 5 dataset contains approximately 24 percent of the samples anticipated to be available once the dataset is complete.

In samples, at EPs, and at systems where PFOA and/or PFOS were reported present, one or more Hazard Index contaminant was reported at or above the minimum reporting level about 68, 70, and 76 percent of the time, respectively. As UCMR 5 monitoring continues, it is possible that additional systems from this subset will report the presence of PFOA, PFOS or a Hazard Index PFAS. The percentage of systems detecting neither PFOA, PFOS, nor a Hazard Index PFAS would then decrease. Table 20 shows the number of Hazard Index PFAS that were observed in samples, at EPs, and at systems where PFOA and/or PFOS were reported.

Table 20: Preliminary UCMR 5 Dataset[1] – Sample, EP, and System Counts According Number of Hazard Index PFAS Reported Present for Systems Where PFOS and/or PFOA were Reported

HI Observed	Samples	EPs	Systems
0	498 (32.1%)	317 (29.7%)	145 (24.2%)
1	573 (37.0%)	403 (37.8%)	223 (37.2%)
2	453 (29.2%)	329 (30.8%)	214 (35.7%)
3	25 (1.6%)	18 (1.7%)	18 (3.0%)
4	0 (0.0%)	0 (0.0%)	0 (0.0%)
Total	1,549	1,067	600

Notes:

[1] The preliminary UCMR 5 dataset contains approximately 24 percent of the samples anticipated to be available once the dataset is complete.

At systems where Hazard Index PFAS were reported in addition to PFOA/PFOS, about 51.0 percent of systems reported multiple Hazard Index PFAS. As described above, it is possible that systems may detect additional PFAS as sample collection continues under UCMR 5. System-level pairwise odds ratios based on the first release of UCMR 5 data are shown in Table 21.

Table 21: Preliminary UCMR 5 Dataset[1] – System-level Counts of Pairwise Chemical Occurrence and Odds Ratios Calculated from Aggregated State Dataset PFAS Samples for PFOA, PFOS, and Hazard Index PFAS

Chem A	Chem B	Chems A and B Reported	Only Chem B Reported	Only Chem A Reported	Neither Chem Reported	Odds Ratio [95% CI]
HFPO-DA	PFBS	10	560	7	3,143	8.0 [3.1-20.5]
HFPO-DA	PFHxS	3	371	14	3,333	1.9 [0.6-6.3]
HFPO-DA	PFNA	0	26	17	3,679	0.0 [0.0-32.6]
HFPO-DA	PFOA	12	417	5	3,286	18.9 [6.9-51.8]
HFPO-DA	PFOS	13	464	4	3,239	22.7 [7.7-66.4]
PFBS	PFHxS	259	115	311	3,034	22.0 [17.1-28.2]
PFBS	PFNA	19	7	551	3,143	15.5 [6.6-36.1]
PFBS	PFOA	290	139	280	3,011	22.4 [17.7-28.4]
PFBS	PFOS	327	150	243	2,999	26.9 [21.3-34.0]
PFHxS	PFNA	17	9	357	3,338	17.7 [8.0-39.2]
PFHxS	PFOA	204	225	170	3,120	16.6 [13.0-21.2]
PFHxS	PFOS	273	204	101	3,142	41.6 [31.8-54.5]
PFNA	PFOA	22	407	4	3,287	44.4 [15.9-123.9]
PFNA	PFOS	20	457	6	3,237	23.6 [9.7-57.4]
PFOA	PFOS	306	171	123	3,119	45.4 [35.0-58.9]

Notes:

[1] The preliminary UCMR 5 dataset contains approximately 24 percent of the samples anticipated to be available once the dataset is complete.

Except for two chemical pairings with HFPO–DA, each pairwise odds ratio estimate between PFAS is statistically significantly greater than one. As previously described, this indicates an increased likelihood of reporting one chemical given that the other chemical is known to be present. HFPO–DA odds ratios with PFBS, PFOS, and PFOA were also statistically significantly above 1. Given that the UCMR 5 dataset is not complete, it is important to note that, for chemical pairs where very few systems have fallen into one or more of the categories of chemical pairings, subsequent sampling may result in substantial shifts in the odds ratio estimate and the associated CI. For example, if one more system reported both HFPO–DA and PFHxS, the odds ratio estimate would increase by 33 percent. On the other hand, if one more system detected both PFOA and PFOS, the odds ratio estimate would shift by less than 1 percent. As the count of systems in each category increases, the odds ratio estimate becomes more stable with subsequent sampling. This may be particularly relevant for relationships with HFPO–DA and other Hazard Index PFAS, given the relatively low number of systems (17 systems) that reported HFPO–DA at or above the minimum reporting level in the preliminary UCMR 5 dataset as of February 2024.

After the release of approximately 24 percent of the data that will be available in the full UCMR 5 dataset, there appears to be considerable PFAS occurrence and co-occurrence demonstrated (USEPA, 2024n). Over 15 percent of systems with appropriate data described above have observed a sample-level exceedance of any of the MCLs while over 9 percent of systems have had an EP with a mean concentration exceeding an MCL. Approximately 75 percent of systems that reported the presence of PFOA or PFOS also observed at least one Hazard Index contaminant. Over half of these systems reported the presence of multiple Hazard Index contaminants. The national PFAS occurrence model estimated between about 6.2 percent and 10.1 percent of all CWS and NTNCWS would have an exceedance of an MCL. The 9.4 percent of UCMR 5 systems that had an EP mean concentration over an MCL is not a direct comparison to this because not all EPs have sampled a year worth of quarterly data and because large systems make up a larger fraction of UCMR systems than systems in the national inventory (the model estimated generally higher concentrations at larger systems). However, separating these UCMR 5 results by system size and weighting according to system counts in the national inventory of systems would result in an estimation of 7.8 percent of all systems having an EP with a mean concentration exceeding an MCL threshold. These estimates are likely to shift as UCMR 5 sampling continues and system sampling regimes are completed.

VII. Analytical Methods

A. Analytical Methods and Practical Quantitation Levels (PQLs) for Regulated PFAS

1. Proposal

The agency proposed two EPA methods to support the monitoring requirements of this regulation. The EPA developed the two liquid chromatography/tandem mass spectrometry (LC/MS/MS) analytical methods to quantitatively monitor drinking water for targeted PFAS: EPA Method 533 (USEPA, 2019b) and EPA Method 537.1, Version 2.0 (USEPA, 2020c). The agency found that all six PFAS proposed for regulation can be measured by both EPA Methods 533 and 537.1, ver. 2.0 and both methods are acceptable for meeting the monitoring requirements of this regulation.

Additionally, the EPA proposed PQLs for the six PFAS proposed for regulation, as outlined in Table 22.

Table 22: PQLs for Regulated PFAS

Contaminant	PQL (ng/L)
PFOA	4.0
PFOS	4.0
HFPO-DA	5.0
PFHxS	3.0
PFNA	4.0
PFBS	3.0

In the proposed rule preamble (USEPA, 2023f), the EPA discussed laboratory performance in the EPA's Unregulated Contaminant Monitoring Rule (UCMR) 5 Laboratory Approval Program (LAP) and found that the UCMR 5 minimum reporting levels are appropriate as the basis for the practical quantitation level (PQL) in this rule. These quantitation levels account for the measurement precision and accuracy that the EPA estimates can be achieved across laboratories nationwide.

2. Summary of Major Public Comments and EPA Responses

Several commenters note analytical differences between EPA Methods 533 and 537.1 such as differences in the quality control (QC) acceptance levels between the methods, sample preservation and holding times, as well as variability in sample and spike duplicates. In some instances, these commenters request specific modification to the methods, revisions to the EPA laboratory certification manual, or for the agency to develop guidance that laboratories and state accreditation/certification bodies could use. These commenters note that while both methods are valid under the proposed rule, variability between the two may lead to differences in sampling results and may impact a water system's compliance status. The EPA agrees that Methods 533 and 537.1 have some differences that allow for analysis of varying chain lengths and molecular structures of PFAS. Method 533 generally captures ''short chain'' PFAS (*i.e.,* those with carbon chain lengths of 4 to 12) and fluorotelomer sulfonic acids. Method 537.1 includes some overlap with Method 533's analyte list while including some longer-chain PFAS. However, the agency notes that all six PFAS proposed for regulation can be analyzed by either Method 533 or 537.1 and neither method has inherent QC issues that lead to significant variation in sampling results when followed. While there are differences between the methods and how they measure their respective target analytes, both EPA Methods 533 and 537.1 perform comparably. The methods are clear and outline specific instructions regarding requirements that are needed for compliance monitoring measurements.

Some public commenters suggested that the EPA allow alternate analytical procedures or modifications to the two published EPA methods for meeting the monitoring requirements in the final rule. The EPA continues to specify the use of Methods 533 and 537.1 because consistent, reliable compliance data are necessary for implementation of the regulation at the maximum contaminant level (MCL) However, the EPA recognizes that improvements in analytical technology and methodology occur. The EPA's Drinking Water Alternate Test Procedure (ATP) Program provides a mechanism for submission and review of alternative methods to measure a contaminant for nationwide use under 40 CFR 141.27. A method developer may apply for the EPA review of a method modification or a new method through the ATP Program. In the meantime, the agency has concluded that Methods 533 and 537.1 are reliable for use in compliance monitoring with respect to accuracy and recovery (lack of bias) and precision (good reproducibility) at the MCL levels.

Several commenters requested that all laboratories be required to identify their quantitation limits (*i.e.,* the smallest detectable concentration of an analyte greater than the detection limit where the accuracy (precision and bias) achieves the objectives of the intended purpose) and/or method detection limits (*i.e.,* the minimum result which can be reliably discriminated from a blank). Specifically, some commenters note if labs have to demonstrate they can get below the PQL, the EPA should establish reporting or detection limits demonstrating they can get to these levels. The EPA is finalizing rule trigger levels below the PQL to support the monitoring provisions discussed in section VIII of this preamble. The EPA disagrees with these commenters that such reporting is needed to support compliance monitoring for the rule and that such reporting would be a cost burden on laboratories. All labs are required per the approved methods to demonstrate whether laboratory reagent blank (LRB) QC samples have background concentrations of less than one-third the minimum reporting level (*i.e.,* the minimum concentration that can be reported as a quantitated value for a method analyte in a sample following analysis). Therefore, for a laboratory to be compliant with the methods, they must be able to detect, not necessarily quantify, analytes at or above ⅓ the minimum reporting level.

Some commenters sought clarity on which methods are approved for use in compliance monitoring for the final PFAS National Primary Drinking Water Regulation (NPDWR). Some of these commenters requested that only Method 533 be approved for monitoring under the final NPDWR, noting that it may be more suitable should additional PFAS analytes within its scope be targeted for regulation at the future date. Others requested that they be permitted to use Method 537, version 1.1. The EPA disagrees and reaffirms that Methods 537.1, version 2.0 and Method 533 are both applicable and suitable for use in compliance monitoring in the final rule. The EPA notes that HFPO–DA is one of the PFAS regulated under this action and only Method 537.1, version 1.0 and version 2.0, and Method 533 support the collection of data for HFPO–DA. The agency notes that the primary difference between Method 537.1, version 1.0 and Method 537.1, version 2.0 is the field reagent blank (FRB) preparation: version 2.0 exposes the FRB to the preservative (Trizma) at the time of field sample collection. Version 1.0 combines the lab reagent water and the preservative together in the FRB prior to field sampling. Version 2.0 was created to more-closely mimic the FRB process used in Method 533. Additionally, Version 2.0 explicitly states that the solid phase extraction (SPE) cartridge sorbents may not be modified with monomers other than styrene divinylbenzene (SDVB).

A few commenters critiqued how the proposed PQLs were established for the rule. Some of these commenters provided feedback on the feasibility of the proposed PQL and suggested that it may be too low, resulting in recurring QC failures that will necessitate repeat sample analysis, increased cost, and reduced laboratory capacity. Other commenters suggest that lower PQLs can be attainable by larger labs with advanced analytical instruments. The agency disagrees that PQLs should be established at either a higher or lower level than that proposed. As discussed in the proposed rule preamble, the PQLs are based on a multi-laboratory assessment of analytical capacity. The EPA derives PQLs which reflect the level that can be reliably quantified within specific limits of precision and accuracy during routine laboratory operating conditions. Based on the multi-laboratory data acquired for the UCMR 5 rule, the EPA has defined the PQL for the PFAS regulated in this rule (Table 22). This quantitation level considers the precision and accuracy that the EPA estimates can be achieved across laboratories nationwide. The EPA anticipates that over time, as technology advances and as laboratories gain experience with the PFAS Methods, laboratories will generally improve their capability to measure at lower levels.

3. Final Rule

The EPA is establishing the following approved methods for use in compliance monitoring in the final PFAS NPDWR: EPA Method 533 (USEPA, 2019b) and EPA Method 537.1, Version 2.0 (USEPA, 2009b; USEPA, 2020c). The PFAS addressed by this regulation can be measured by both EPA Methods 533 and 537.1 and either method is acceptable for meeting the monitoring requirements of this regulation. Table 1 to paragraph (f)(1)(iv) of § 141.903 of subpart Z lists the PQLs for the PFAS regulated under this action.

VIII. Monitoring and Compliance Requirements

A. What are the Monitoring Requirements?

1. Proposal

The EPA proposed requirements for community water systems (CWS) and non-transient non-community water systems (NTNCWSs) to monitor for six PFAS. The agency proposed to amend 40 CFR part 141 by adding a new subpart to incorporate the regulated PFAS discussed in this preamble. Under this new subpart, public water systems (PWSs) would be required to sample EP using a monitoring regime based on the EPA's Standard Monitoring Framework (SMF) for Synthetic Organic Contaminants (SOCs).

The EPA proposed the following requirements for initial monitoring, which systems would be required to complete by the date three years after the date of rule promulgation (see section VIII.F of this preamble for more information). The EPA proposed that, consistent with the SMF for SOCs, groundwater systems serving greater than 10,000 persons and all surface water systems would be initially required to monitor quarterly within a 12-month period for regulated PFAS. To provide additional flexibilities for small groundwater systems, the EPA proposed to modify the SMF for SOCs such that groundwater systems serving 10,000 or fewer persons would be initially required to monitor only twice for regulated PFAS within a 12-month period, each sample at least 90 days apart. In the proposal, all systems would be allowed to use previously acquired monitoring data to satisfy the initial monitoring requirements (see section VIII.C of this preamble for additional details about using previously acquired monitoring data to satisfy initial monitoring requirements). Based on the SMF, the EPA also proposed that primacy agencies be able to use initial monitoring results to reduce compliance monitoring frequency for a system to once or twice every three years (depending on system size) if the monitoring results are below the proposed rule trigger level (defined in the following paragraphs).

The EPA proposed that, after initial monitoring, water systems would conduct compliance monitoring to demonstrate that finished drinking water does not exceed the maximum contaminant levels (MCLs) for regulated PFAS. The EPA proposed that systems with multiple EP may establish different compliance monitoring schedules for those EP depending on their monitoring results.

The EPA proposed to base compliance monitoring requirements on initial monitoring results and on system size. Then subsequent monitoring requirements would be based on results from compliance monitoring and, for systems on triennial monitoring, also on system size. To determine compliance monitoring frequency only, the EPA proposed a rule trigger level of one-third the MCLs (1.3 ng/L for PFOA and PFOS and 0.33 for Hazard Index PFAS (PFHxS, PFNA, HFPO–DA, and PFBS)). If results for an EP are below the trigger level, systems would be eligible for reduced monitoring. To implement this provision, the EPA proposed to include the ‘‘trigger level’’ concept in the new subpart.

As proposed, each water system would be eligible for reduced compliance monitoring at each EP for which all PFAS results are below the rule trigger level, according to the following schedule:

- A water system that serves 3,300 or fewer customers would be required to analyze one sample for all regulated PFAS per three-year compliance period at each EP where the water system does not have results for any regulated PFAS at or above the rule trigger level (1.3 ng/L for PFOA and PFOS and 0.33 for the Hazard Index PFAS (PFHxS, PFNA, HFPO–DA, and PFBS)),
- A water system that serves more than 3,300 persons would be required to analyze two samples for all regulated PFAS at least 90 days apart in one calendar year per three-year compliance period at each EP where the water system does not have results for any regulated PFAS at or above the rule trigger level (1.3 ng/L for PFOA and PFOS and 0.33 for the Hazard Index PFAS (PFHxS, PFNA, HFPO–DA, and PFBS).

In the proposal, if any result for an EP is at or above the rule trigger level for regulated PFAS, the water system would be required to monitor at that EP for all regulated PFAS quarterly. For compliance monitoring collection schedules, the EPA did not specify the required number of days between sampling events and only required collection during a quarter. Systems monitoring an EP less frequently than quarterly whose sample result is at or above the rule trigger level would also be required to begin quarterly sampling at the EP where regulated PFAS were observed at or above the trigger level. In either case, the primacy agency would be able to allow a system to move an individual EP to a reduced monitoring frequency when the primacy agency determines that the EP is below the rule trigger level and reliably and consistently below the MCL. However, primacy agencies would not be permitted to determine that the EP is below the rule trigger level and reliably and consistently below the MCL until at least four consecutive quarters of quarterly compliance monitoring have occurred with all sample results below the rule trigger level.

Additionally, related to laboratory capacity considerations, the EPA described in the proposal that it anticipates that laboratories will be able to adjust to demand and that the demand will be distributed across the three-year implementation period.

2. Summary of Major Public Comments and EPA Responses

The following discussion details numerous comments the EPA received on the proposed monitoring requirements, both for initial monitoring and long-term compliance monitoring.

The majority of comments the EPA received on the initial monitoring requirements related to the number of initial samples systems would be required to collect and the intervals between required samples. Most commenters were generally supportive of the EPA's proposed initial monitoring requirements, including the flexibilities to use previously acquired monitoring data to satisfy some or all the initial monitoring requirements and, for those groundwater systems serving 10,000 or fewer that do not have this data, that they be required to only collect two samples at each EP to satisfy initial monitoring requirements. For a discussion of comments and final rule requirements specific to the use of previously acquired monitoring data to satisfy the initial monitoring requirements see section VIII.C of this preamble.

While most commenters were supportive of the number of initial monitoring samples the EPA proposed, a few commenters indicated they thought the EPA should not allow the flexibility for groundwater systems serving 10,000 or fewer to collect only two samples and instead require quarterly samples be collected by all systems to meet initial monitoring requirements, which would be fully consistent with the SMF framework for other SOCs. A couple of these commenters suggested that there are no data demonstrating that smaller systems are less likely to have elevated levels of PFAS than large systems or that groundwater systems are less likely to have elevated levels of PFAS than surface water systems. Additionally, other commenters generally suggested that two samples may not generate enough data to accurately capture the level of PFAS in drinking water and any potential seasonal variability. Related to potential seasonal changes in measured PFAS concentrations, some commenters from state agencies indicated that they have not observed seasonal variations in concentrations of PFAS measured by groundwater systems, whereas other commenters suggested the opposite and that they have seen changes seasonally based on their state's monitoring data.

The EPA disagrees with commenters that suggest two samples for small groundwater systems would not accurately capture the baseline level of regulated PFAS in drinking water. The EPA determined the initial monitoring requirements based on both source water type and system size considerations. First, from a national-level perspective, the EPA's model for estimating national PFAS drinking water occurrence (see section VI.E of this preamble) indicates that, regardless of source water type, small systems generally have lower mean PFAS concentrations and lower within-system variability than large systems. Further accounting for source water type, as compared to all groundwater systems, all surface water systems potentially have a larger number of sources of contamination and greater hydrology variability so more monitoring data is necessary to ensure an appropriately protective monitoring schedule. Both the differences in the occurrence estimations for large and small sized systems as well as the general source water characteristics of groundwater systems were collectively considered as part of establishing the proposed initial monitoring requirements for small groundwater systems. Consequently, the agency expects that small groundwater systems would be less likely to experience variations throughout a year and, where there may be seasonal variations, requiring the samples to be collected in different parts of a year would provide sufficient information to determine the appropriate compliance monitoring schedule. Furthermore, given the different experiences cited by commenters, possible seasonal variation is likely based on the specific geographic location and other localized factors. If there are regional factors that suggest more frequent sampling is warranted, the rule provides that primacy agencies may increase the required monitoring frequency, where necessary, to detect variations within the system (*e.g.,* fluctuations in concentrations due to seasonal use or changes in water source).

In response to comments about the alignment of Unregulated Contaminant Monitoring Rule (UCMR) 5 sampling with initial monitoring requirements, a couple of commenters indicated that requiring larger groundwater systems to collect four samples would translate into these systems needing to collect two additional samples beyond those collected for the UCMR 5 monitoring effort. The EPA acknowledges that while the initial monitoring requirements generally align with the UCMR 5 sampling requirements, groundwater systems serving greater than 10,000 would need to collect two additional samples and notes that they have the three years following rule promulgation to complete this monitoring. As described previously, the model for estimating national PFAS drinking water occurrence indicates that larger systems have greater within-system variability than smaller systems, therefore it is appropriate that these larger groundwater systems collect four initial monitoring samples; this is consistent with initial monitoring requirements for groundwater systems under existing SOC National Primary Drinking Water Regulations (NPDWRs).

In addition, a couple of commenters recommended that the number of required samples for initial monitoring be based on the results of the first two samples, with subsequent monitoring only required if regulated PFAS are detected in those earlier samples. The EPA recognizes there is some logic to this approach; however, there would be challenges implementing it. Specifically, it could be challenging for primacy agencies to track and implement the proposed approach, particularly for groundwater systems serving 10,000 or fewer which would require the additional samples to occur in quarters not represented by the first two samples. Furthermore, tracking this varying monitoring would result in additional administrative burden and oversight challenges for primacy agencies, rather than having a consistently defined schedule for monitoring requirements as is used for other SOCs.

The EPA also received several comments from state agencies about the required intervals associated with initial quarterly and semiannual sample collection. In its proposal, the EPA specified that samples be collected at least 90 days apart, whether the samples were required of a system monitoring on a quarterly basis or a system monitoring semi-annually. A couple of commenters noted that they believed that semiannual samples should be separated by more than 90 days to better capture seasonal variations (*e.g.,* seasonal changes in the percent contributions of water blended from different sources, other fluctuations in concentrations). One commenter suggested semiannual samples should be collected at least 180 days apart, which would also be in better alignment with the required schedule for UCMR 5 semiannual sampling. The EPA agrees with these comments. In the final rule, the EPA is requiring that the samples be collected 5 to 7 months apart for semiannual initial monitoring (see table 2 to paragraph (a)(4)(i)(B) of the regulations governing the UCMR program in 40 CFR 141.40).

With respect to the sample collection timing requirements for quarterly initial monitoring (for all surface water systems and groundwater systems serving greater than 10,000), a few commenters indicated that they were opposed to the proposed requirement for samples to be spaced at least 90 days apart. These commenters indicated that such a requirement was unnecessarily prescriptive and would make sample collection logistically challenging for public water systems. These commenters suggested the EPA change the required spacing in a way that still satisfies the EPA's intent to not have samples collected only a few days apart, but in different quarters, so that quarterly samples are more representative of fluctuations in concentrations over time. The EPA agrees with these comments and sees the value of systems being able to use four existing samples collected in separate quarters but also allow flexibility that they are not all spaced at least 90 days apart. In the final rule, the EPA is modifying the required spacing of quarterly initial monitoring samples to be 2 to 4 months apart if samples are collected in a 12-month period. For systems that would need to supplement previously acquired data to satisfy all the initial monitoring requirements, the final rule requires that they must also be 2 to 4 months apart from the months of available pre-existing data. This will also better parallel the language outlining the required spacing of quarterly samples collected for the UCMR 5 monitoring effort.

Some commenters asked the EPA to clarify which systems would be subject to the initial monitoring requirements for surface water systems and which systems would be subject to the requirements for groundwater systems, in some cases presenting examples of specific scenarios. One example is when a system relies on surface water at some EP and groundwater at other EP. The EPA has modified the language of the final rule in § 141.902(b)(1)(ii) to clarify that initial monitoring requirements are to be determined based on the type(s) of water serving as the source for a given EP; thus, one system may have different initial monitoring requirements that apply to different EP. In response to questions, the EPA is clarifying in § 141.902(b)(1)(iv) that, if an EP uses water blended from multiple sources (some groundwater and some surface water), or if it uses different types of sources throughout the year, the system must follow the monitoring frequency for a surface water system (since water from surface water sources is used at least in part, for at least a portion of the year). This approach is more protective of public health because, as described earlier, generally surface water systems have more variable hydrology and potentially more sources of contamination so more monitoring data is necessary to ensure an appropriately protective monitoring schedule.

A couple of commenters asked for clarification about whether EP supplying groundwater under the direct influence of surface water (GWUDI) would qualify for semiannual initial monitoring. As noted in § 141.902(b)(1)(iii), GWUDI systems follow the requirements for surface water systems. GWUDI systems may be as susceptible to contamination as surface water systems; thus, these systems must use the sampling requirements for surface water during the initial sampling phase to establish baseline levels of regulated PFAS.

Regarding the requirements for longer-term compliance monitoring, the comments the EPA received related primarily to the frequency with which sampling would occur under different circumstances, whether each EP would be allowed to be on a different compliance monitoring schedule, and the trigger levels that would support decisions about reduced triennial monitoring. Regarding the latter point, commenters also addressed laboratory capabilities to measure levels below practical quantitation levels (PQLs).

The EPA's proposal would allow systems eligible for reduced monitoring, and serving 3,300 or fewer, to collect one sample triennially and would allow eligible larger systems to collect two samples during a three-year compliance period. The EPA specifically requested comment on whether all water systems, regardless of system size, should be allowed to collect and analyze one sample per three-year compliance period if the system does not measure any regulated PFAS in their system at or above the rule trigger level. A few commenters stated that they did not agree with a different number of triennial samples eligible systems must collect based on the size of the population a system serves. These commenters indicated that they believe that one sample collected every three years is sufficient for systems of any size on reduced monitoring. The EPA agrees with these commenters that systems eligible for triennial monitoring should be allowed to collect one sample every three years, regardless of system size, especially considering other changes to the compliance monitoring framework, as described subsequently.

Several commenters recommended that an annual sampling frequency tier be added to the required monitoring framework for various reasons including the mobility and persistence of PFAS in the environment, to ensure that systems that have demonstrated elevated levels of regulated PFAS are not allowed to move directly from quarterly to triennial monitoring, and based on their concerns that some laboratories may not be able to produce results at or below the rule trigger levels (resulting in some systems remaining on quarterly monitoring indefinitely even if they can consistently demonstrate they are below the MCLs). A few commenters supported offering three possible monitoring frequencies: quarterly, annually, and triennially, whereas many other commenters recommended against allowing triennial sampling at all and recommended that sampling be required no less than annually, to best protect public health. Those commenters supportive of allowing both annual and triennial monitoring, depending on prior sample results, suggested that annual monitoring should be an option for systems with regulated PFAS concentrations that are reliably and consistently below the MCLs. This modification would parallel the three tiers of monitoring allowed for other organic chemicals under the SMF.

The EPA does not agree with the comments suggesting that no systems should be allowed to sample triennially and that the longest sampling interval at any location should be one year. Based on the EPA's national occurrence estimates, most water systems subject to the rule's requirements will not have results for regulated PFAS that exceed the MCLs, and many will not identify PFAS at or above the triggers for reduced monitoring. These systems, after demonstrating results below the trigger level and therefore no or very little presence of regulated PFAS during the initial monitoring period or through ongoing compliance monitoring, should be able to reduce their monitoring burden and conduct triennial sampling. These monitoring requirements will sufficiently maintain public health protection. If a system monitoring triennially did have a sample result with elevated levels of a regulated PFAS (at or above the trigger level), it would be required to immediately initiate quarterly monitoring. Additionally, the rule specifically provides that primacy agencies may increase the required monitoring frequency for compliance sampling for a variety of reasons, including to detect variations within specific systems (*e.g.,* fluctuations in concentrations due to seasonal use patterns or changes in water sources).

For any system that has regulated PFAS concentrations at or above the trigger level, but reliably and consistently below the applicable MCL, the EPA is introducing in the final rule an annual monitoring frequency within the compliance monitoring framework, consistent with the SMF for SOCs. A demonstration of reliably and consistently below the MCL would include consideration of at least four quarterly samples below the MCL. Annual samples would be collected during the quarter with the highest concentration measured during the prior round of quarterly sampling. The EPA expects this modification in the final rule to reduce the number of systems that are required to be on quarterly monitoring for extended periods of time, compared to the EPA's proposal.

In adopting a three-tiered monitoring framework, the EPA is modifying the required sampling frequency from triennial to annual for systems determined by states to be reliably and consistently below the MCL and changing the threshold for this determination from the trigger level to the MCL. To further reduce monitoring, any system that transitions into annual sampling will be required to collect three years of annual samples each of which show concentrations of regulated PFAS below trigger levels (*i.e.,* not an average of the three annual sample results) before then being eligible for triennial monitoring. Moreover, no system required to collect quarterly samples during compliance monitoring would be allowed to transition to triennial monitoring without first conducting three years of annual monitoring, with all results below the trigger level. If eligible for triennial monitoring, the sample collected triennially would need to be collected in the same quarter during which prior results were highest.

This additional tier is intended to create a gradual step-down schedule for affected EP to confirm levels of regulated PFAS are remaining consistently low or decreasing. The modifications to the requirements for a reliable and consistent determination and the creation of the new annual sampling tier in the final rule make the requirements for regulated PFAS more consistent with the NPDWR requirements for SOCs. They also represent flexibilities that address concerns about laboratory capability concerns. The EPA believes this three-tier approach, including the eligibility criteria for each outlined above, provides the best approach to protect public health and moderate the total cost of sampling borne by a system.

The EPA also received a few comments about the practice by systems that have installed treatment for PFAS to regularly sample finished water to ensure the efficacy of their treatment media (*e.g.,* filters), above and beyond what they would do for compliance monitoring. A few commenters suggested systems that have installed treatment would conduct this additional sampling voluntarily, typically for process control purposes. A few state agency commenters suggested that any system that is treating its water for PFAS should be required to sample more frequently than triennially (*e.g.,* annually) no matter the levels of previous PFAS detections, since the effectiveness of treatment media may decline over time, if not replaced. The EPA disagrees with the commenters recommending a greater sampling frequency for systems that treat their water for PFAS and does not see a compelling reason to depart from the three-tier compliance monitoring program for a system that has installed treatment. In the final rule, the EPA is adding an annual tier of sampling for any system with concentrations reliably and consistently below the MCL but not consistently below the trigger level. The EPA believes this tier will likely apply to most systems treating their water for regulated PFAS, at least for the first three years of treatment, as the EPA estimates as part of its rule costs that systems needing to install treatment will assume a treatment target of 80 percent of the MCLs. The majority of systems with elevated levels of regulated PFAS contamination are likely to sample quarterly, at least initially (unless they have treatment for PFAS in place prior to the collection of initial monitoring samples). In practice, the result is that most systems with PFAS contamination will likely not be eligible for triennial sampling unless their PFAS treatment is consistently optimized and maintained. However, the rule provides that primacy agencies may increase the required monitoring frequency, where necessary to detect variations within the system, and this approach could be applied to those systems that have installed treatment. In addition, the EPA notes that, when systems are treating for other regulated chemicals pursuant to NPDWRs, no distinctions are made between the monitoring frequency required of a system that is treating for a chemical and a system that has not installed treatment. Thus, not establishing a different monitoring frequency specifically for systems that are treating their water for PFAS is consistent with existing NPDWRs.

The EPA requested comment on the proposed allowance of a water system to potentially have each EP on a different compliance monitoring schedule based on specific EP sampling results (*i.e.,* some EP being sampled quarterly and other EP sampled only once or twice during each three-year compliance period), or if compliance monitoring frequency should be consistent across all of a system's sampling points. A few commenters recommended that all EP used by a system monitor at the same frequency, or that doing so be optional, to reduce the complexity of monitoring requirements or the potential for mistakes to be made with respect to sampling windows. However, the overwhelming majority of those who commented on this topic indicated they supported allowing different sampling frequencies for different EP. The EPA agrees that it would be beneficial to allow different sampling frequencies for different EP because it would allow utilities to realize cost savings if only the EP with elevated levels of PFAS are required to sample most frequently. In addition, the EPA notes it allows systems to use different sampling frequencies for different EP for compliance with other NPDWRs.

The EPA requested comment on monitoring-related flexibilities that should be considered to further reduce burden while also maintaining public health protection, including setting a rule trigger level at different values than the proposed values of 1.3 ng/L for PFOA and PFOS and 0.33 for the Hazard Index PFAS (PFHxS, PFNA, HFPO–DA, and PFBS). Alternative values of 2.0 ng/L for PFOA and PFOS and 0.50 for the Hazard Index PFAS were identified as possibilities. The EPA received numerous comments on the proposed rule trigger levels. Comments addressed the proposed values, specifically for PFOA and PFOS, and their intended purpose for determination of compliance monitoring frequency. Several commenters suggested that the proposed values (*i.e.,* 1.3 ng/L for PFOA and PFOS and 0.33 for the Hazard Index) are too high and the EPA should instead set lower trigger level to ensure greater public health protection. Many other commenters suggested the opposite, stating that the proposed levels are too low, that laboratories will not be able to achieve these levels, and that it may exacerbate any laboratory capacity issues. Consequently, some of these commenters were concerned that water systems would be ineligible for reduced monitoring based on their laboratory's analytical limitations. Several commenters suggested that the proposed values are inconsistent with the SMF for SOCs.

Many who commented on the subject were fully supportive of the EPA's proposed alternative trigger level values of 2.0 ng/L for PFOA and PFOS and 0.50 for the Hazard Index, while others expressed support for the inclusion of trigger levels only if these higher levels were incorporated. Some noted that these higher trigger levels would better align with current laboratory capabilities and allow greater use of previously collected drinking water data (to demonstrate systems are eligible for reduced triennial monitoring under the rule's initial monitoring requirements). A few commenters recommended alternative values of 70–80 percent of the MCLs be used as the trigger levels.

The EPA agrees with commenters that the trigger levels should be finalized as one-half of the MCLs (*i.e.,* PFOA and PFOS at 2.0 ng/L each, PFHxS, PFNA, and HFPO–DA at 5 ng/L each, and Hazard Index at 0.5). Using data submitted as part of the UCMR 5 LAP as a reference point, the EPA notes that 47 of 53 laboratories (89 percent) that applied for UCMR 5 approval generated a minimum reporting level confirmation at 2 ng/L (one-half the proposed MCL) or less for Method 533. This suggests that most laboratories with the necessary instrumentation to support PFAS monitoring have the capability to provide screening measurement results at the revised trigger level of one-half of the MCL. This corresponds with other comments described in section VIII.C of this preamble that provided their experience that laboratories are capable of reliably quantifying values below the PQLs, particularly to 2.0 ng/L for PFOA and PFOS.

Additionally, based on the EPA's evaluation of state drinking water data, updating the final rule trigger levels (to one-half of the MCL) will result in a considerable number of additional water systems significantly reducing their ongoing monitoring frequency from quarterly or annual monitoring to triennial monitoring. Although this modification from one-third of the MCL to one-half of the MCLs may provide slightly less information on a water system's measured PFAS levels as a result of their less frequent monitoring, the trigger levels for the final rule (*i.e.,* one-half of the MCLs) will ensure sufficient public health protection while reducing burden for water systems.

Many other commenters stated that either trigger levels should be removed from the rule entirely or that trigger levels should not be set to any levels below PQLs since these represent the level that can be reliably measured with a high degree of precision and accuracy across all laboratories. Several of these commenters suggested that data below the PQL are unreliable, would result in higher costs, and should not be used as the basis for any regulatory decisions. Thus, they suggested that if trigger levels are incorporated, they should be the same as the PQLs. These commenters also cited laboratory challenges in achieving measurement below the PQLs and suggested that water systems would not be eligible for reduced triennial monitoring as a result of these limitations. Additionally, some of these commenters suggested that decision making based on any values below the PQLs may exacerbate laboratory capacity issues, claiming that such trigger levels would result in errors, such as false positives, which would lead to increased monitoring where samples need to be re-tested.

The EPA emphasizes that the use of trigger levels set at values below the MCLs is consistent with other SOCs under the SMF and not novel for drinking water regulations (as described in the subsequent paragraph). Their use allows water systems the opportunity to reduce their monitoring schedule and burden where it can be demonstrated through sampling results that they are at low risk of PFAS contamination. In the absence of trigger levels, or some other threshold, all water systems would be deprived of the opportunity for reduced monitoring. At a national level, were the EPA to eliminate reduced monitoring options, this would result in a significant increase in costs to utilities. Consequently, the EPA is choosing to incorporate these levels to allow flexibility and reduce burden for water systems while maintaining health protection.

For commenters that suggest the trigger levels should be identical to the PQLs, particularly for PFOA and PFOS, the EPA disagrees as the agency must have greater assurance that the levels are below the regulatory standard, the systems are actually lower risk, and a reduced monitoring schedule is appropriate. Specifically, in the case of PFOA and PFOS, the EPA believes it would represent an unacceptable public health risk to set trigger levels at the PQLs because the EPA is setting the MCL at the PQL which means that it represents the ''maximum permissible level.'' Moreover, the approach of considering measured levels lower than PQLs for determining monitoring frequency is not novel but has been part of the drinking water standards for many years. Many drinking water standards even use a method detection limit, which by definition is lower than the PQL. Under the SMF for SOCs, for example, results both at or below detection limits and between detection limits and the MCL are utilized for monitoring frequency determination. Additionally, 40 CFR 141.24(h)(7) prescribes the monitoring frequency for organic contaminants based on sample results relative to detection limits (as defined in in paragraph (h)(18) of the same section). In each of these cases, detection limits are below their PQLs (often by a factor of 10). The approach in this rule—using levels lower than the PQL to determine monitoring frequency—is consistent with the EPA's approach for other NPDWRs (see section V of this preamble).

As described earlier, some commenters raised concerns about potential laboratory analytical and capacity issues. Some suggested that laboratories cannot achieve levels below the PQLs, which would result in water systems not being eligible for reduced monitoring based on not demonstrating results below trigger levels. The EPA recognizes that some laboratories may not be able to produce results at these lower levels with the same degree of accuracy and precision as results at or above the PQLs, and notes that there is not a requirement that they do so for these purposes. The EPA uses the PQL to inform the MCL feasibility determination and the same level of precision and accuracy is not required to determine monitoring frequency. Along these lines, several commenters questioned if the sample results must be quantified to be used for the determination of monitoring frequency, given the proposed trigger level values were set below the PQLs, requesting further clarity from the EPA on how to interpret and utilize quantified and non-quantified data. Furthermore, some commenters suggested that if values below the PQLs are used, only quantified results should be used for determining monitoring frequency. Other commenters stated there should not be a numerical value associated with results below the PQL (*e.g.,* results between the trigger levels and the PQLs) and instead such results should only be reported on an absence/presence basis.

The EPA agrees that results below the PQL may not have the same precision and accuracy as higher-level measurements; however, results below the PQL can be sufficiently determined for these purposes. Data below the PQL will be critical to ensuring that systems are monitoring at the correct frequency and whether a contaminant is present within a certain range. Moreover, while results near the trigger level may be less definitive than results at or above the PQL, such results are appropriate for establishing monitoring frequency, as well as for reporting as part of the annual Consumer Confidence Report (CCR). CCR reporting is based on *detected* contaminants and for the purposes of the PFAS NPDWR, § 141.151(d) defines ''detections'' as results at or above the rule trigger levels (see section IX of this preamble for more information on CCR requirements).

Under this final rule, for monitoring frequency determination purposes, systems are required to use all compliance sample results, including those below the PQLs and not quantified with the same precision and accuracy as is associated with the MCL compliance calculation determination. Additionally, the determination of monitoring frequency is not based on a running annual average result, but each individual sampling result. As an illustration of the approach, if a water system has quarterly sampling results at an EP from initial monitoring for PFOA that are 2.0, 1.5, 5.0, and 1.5 ng/L, there are two results (*i.e.,* 2.0 and 5.0 ng/L) at or above the EPA's final trigger level for PFOA (*i.e.,* 2.0 ng/L). Thus, the water system would not be eligible for triennial monitoring at this EP for all regulated PFAS when compliance monitoring begins. Providing a different example, if a water system that is currently required to conduct quarterly compliance monitoring has quarterly sampling results at an EP for PFOA that are 2.0, 3.5, 2.5, and 1.5 ng/L, all results are below the MCL for PFOA (*i.e.,* 4.0 ng/L), however three results are above the PFOA trigger level. In this case, because four quarters of data have been collected and assuming all other regulated PFAS sampling results are below their MCLs as well, the water system could be deemed reliability and consistently below the MCL by the primacy agency and be eligible to monitor annually at this EP. For all frequencies of ongoing compliance monitoring, including quarterly, annual and triennial, this determination would be done the same where all sample results are used, even those below the PQLs.

Many commenters requested that the EPA provide clarification on how laboratories and PWSs should report levels below the PQLs for monitoring frequency purposes. All results at or above the trigger level are to be reported as numeric values and used for determining monitoring frequency. Under the EPA approved analytical methods discussed in section XII, numeric values as low as the rule trigger levels will be available because of the need to meet ongoing QC requirements of the methods for blanks, demonstrating no background contamination. Within each analytical batch of samples, the laboratory must document passing blank QC criteria by attaining qualitative measurements of the regulated PFAS that are no higher than one-third of the laboratories reporting limit, which must be at or below the PQL. The EPA intends to provide guidance materials with details and examples on this to support successful implementation of the final rule.

Some commenters suggested the potential for confusion related to the differences in how results less than PQLs are used in monitoring frequency determination and the MCL compliance determination. Several commenters suggested that there should be a consistent approach. Most commenters suggested that the approach should follow that of the MCL compliance determination, where zero is used in the calculation of annual averages when measured values are below PQLs. The EPA reiterates that the trigger levels are used for establishing appropriate monitoring frequency. For certain regulated PFAS, they are set at a defined threshold that shows if these PFAS are present or absent. The PQLs, which are used for the MCL compliance determination, are set at specific concentrations that laboratories nationwide can measure with high certainty. To alleviate possible confusion, the EPA intends to provide communication materials on these monitoring requirements to support successful implementation of the final rule. Nevertheless, the difference in approach (between data used for compliance monitoring determinations and data used to determine monitoring frequency) reflects the most appropriate application of the data for each of the intended purposes and assures that adequate monitoring is occurring in systems where the regulated PFAS have been shown to be present at the trigger level or higher. The EPA's rationale is described in detail in section VIII.B of this preamble.

Several other issues related to monitoring flexibilities were raised in public comments. One commenter asked, if one EP has a result for a single regulated PFAS at a concentration above the trigger level, but other regulated PFAS are below trigger levels, must the system initiate quarterly sampling for all regulated PFAS at the EP or are they only required to initiate quarterly sampling for PFAS observed at or above the trigger level. As described in the rule proposal, if a regulated PFAS is detected at or above a trigger level, the system must monitor quarterly at that sampling point for all regulated PFAS. This is appropriate as the same analytical methods are used for the analysis of all regulated PFAS (no extra analyses need to be performed to measure the other PFAS) and the regulated PFAS have been shown to significantly co-occur.

In addition, commenters questioned whether quarterly sampling would be triggered when a result is equal to but does not exceed the trigger level for systems monitoring triennially. One commenter pointed out that the language proposed for inclusion in § 141.905(b)(2) stated that systems monitoring triennially whose sample result is at or exceeds the trigger level must begin quarterly sampling, whereas § 141.902(b)(2)(ii) stated the trigger level must be exceeded before quarterly monitoring is required. The EPA is clarifying this point in the final rule to reflect the EPA's intent that quarterly sampling would be triggered when a result is at or above the trigger level as prescribed in § 141.905(b)(2). This same approach has been used in other NPDWRs (*e.g.,* for SOC trigger levels).

3. Final Rule

This final rule establishes initial monitoring requirements and reflects minor modifications to the proposed approach. Groundwater CWS and NTNCWS serving 10,000 or fewer must collect two (semiannual) samples in a consecutive 12-month period and must collect the samples 5 to 7 months apart, to better capture seasonal variation. Groundwater CWS and NTNCWS serving greater than 10,000 and all surface water CWS and NTNCWS must collect four (quarterly) samples 2 to 4 months apart in a consecutive 12-month period. The EPA is maintaining the provision described in the proposed rule that allows PWSs to use previously collected data to satisfy initial monitoring requirements; see § 141.902(b)(1)(vi). Systems that need to collect additional quarterly samples to meet the initial monitoring requirements may sample outside of a 12-month period, if all quarters are represented with sample months 2 to 4 months apart. This 2-to-4-month interval also aligns with UCMR 5 sampling requirements for surface water systems subject to this rule and better captures possible seasonal variability establishing a well-informed baseline. In addition, the EPA is modifying the proposed initial monitoring requirements to now specify that if the water source for the EP is surface water, a blend of surface water and groundwater, or GWUDI, the initial monitoring requirements for surface water source (4 quarterly samples) apply. If the EP source is only groundwater, initial semiannual monitoring is required.

The EPA is modifying the number of samples required for some systems with sampling locations eligible for triennial monitoring. Regardless of the population served, all systems with sampling locations eligible for triennial sampling will collect one sample every three years. The sample is to be collected during the quarter with the highest prior concentration identified in the most recent year when samples were collected.

In the final rule the EPA is establishing a third tier for monitoring frequencies and updating the proposed requirements for each tier. The new monitoring frequency tier provides for annual monitoring at sampling locations that have collected at least four consecutive quarterly samples following initial monitoring if the primacy agency determines the results at that EP are reliably and consistently below the MCL. In establishing this tier, the EPA is removing the proposed rule requirement for a state to determine that the running annual average (RAA) concentration is below the trigger levels to reach this reliably and consistently below the MCL determination. Instead, in the final rule, reliably and consistently below the MCL means that each of the sample results for the regulated PFAS are below the applicable MCLs. In this new annual monitoring tier, if EP receive the reliably and consistently below the MCL determination and remain below the MCLs in subsequent sampling, even if above a trigger level, they may continue on an annual monitoring schedule.

The criteria eligibility for triennial monitoring have been changed accordingly. EP with all results below the trigger levels during initial monitoring are eligible for triennial monitoring, as described in the proposed rule. But, under the final rule, if an EP is required to conduct quarterly sampling during the compliance monitoring period, then triennial monitoring is only available after the EP has three consecutive annual samples that each contain concentrations below the trigger level. For EP that consistently have results between the trigger levels and the MCLs, as described previously most would remain on annual monitoring, rather than quarterly monitoring, which provides a sufficient indication of contaminant level while reducing the total sampling costs.

With respect to whether different EP for a particular water system may be sampled at different compliance monitoring frequencies, based on specific EP sampling results, the final NPDWR affirms this flexibility, as proposed. In addition, there is no change to the language in the final rule discussing the timing for taking quarterly samples during the long-term compliance monitoring period. The EPA does not specify a required interval between samples; the requirement is quarterly.

The EPA is finalizing rule trigger levels for compliance monitoring frequency purposes only at one-half of the MCLs for regulated PFAS (*i.e.,* 2.0 ng/L for PFOA and PFOS, 5 ng/L for PFHxS, PFNA, and HFPO–DA, and 0.5 for Hazard Index). If all PFAS results for an EP are below these levels, the EP would be eligible for triennial monitoring, with the following exception. If sampling location is under an annual monitoring schedule, it would be eligible for triennial monitoring following three consecutive annual samples with all sample results below the trigger levels.

The EPA's proposed rule included monitoring requirements specific to PFAS. To avoid possible confusion, the EPA is amending 40 CFR 141.24(h) to clarify that the applicable monitoring requirements for PFAS are in 40 CFR 141.902 and that the monitoring requirements for non-PFAS SOCs in 40 CFR 141.24(h) do not apply to PFAS.

B. How are PWS compliance and violations determined?

1. Proposal

Consistent with existing rules for determining compliance with NPDWRs, the EPA proposed that compliance would be determined based on the analytical results obtained at each sampling point. For systems monitoring quarterly, compliance with the proposed MCLs would be determined by calculating RAAs for each sampling point. As proposed, eligibility for reduced monitoring would be determined by the sample result(s) at the sampling point. If the sample result(s) are at or exceed the rule trigger level, the system would be required to revert to quarterly sampling, for all regulated PFAS, at each EP where a result is at or above the trigger level. In such case, the sample event that included a result(s) at or above the trigger level would be considered the first quarter of monitoring in calculating the RAA.

An RAA is calculated using results for samples taken at a particular monitoring location during the previous four consecutive quarters. As proposed, if a system takes more than one compliance sample during each quarter at a particular monitoring location, the system must average all samples taken in the quarter at that location to determine the quarterly average, which would then be used in calculating the RAAs. Conversely, if a system does not collect required samples for a quarter, the RAA would be based on the total number of samples collected for the quarters in which sampling was conducted. As proposed, MCL compliance determinations would not be made until a system has completed one year of quarterly sampling, except in the case where a quarterly sampling result is high enough that it will clearly cause the RAA to exceed an MCL (*i.e.,* the analytical result is greater than four times the MCL). In that case, the system would be in violation with the MCL immediately.

In the proposal, when calculating the RAAs, if a sample result is less than the PQL for the monitored PFAS, the EPA proposed to use zero to calculate the average for compliance purposes.

2. Summary of Major Public Comments and EPA Responses

The agency received a few different types of comments on how the compliance determination and violations were proposed to be assessed. Many commenters supported the EPA's approach to assess violations, including that violations are only assessed through an RAA for systems conducting quarterly monitoring. A couple of commenters suggested that in a scenario where a particular high quarterly sample (*i.e.,* result greater than four times the MCL) would cause the RAA to exceed an MCL, the system should not be deemed out of compliance until the end of the quarter (to allow utilities to conduct additional monitoring during that quarter and average the results from the multiple samples). The EPA disagrees with commenters that suggest additional voluntary sampling be used in calculating the quarterly average. The final rule requires that a compliance sample be taken during each quarter for those systems conducting quarterly monitoring. Further, as prescribed under 141.902(b)(2)(v), the state may require a confirmation sample for any sampling results and, if this sample is required, the result must be averaged with the first sampling results and used for the compliance determination. Therefore, any samples other than a state-required confirmation sample should not be averaged within the quarterly compliance result which will be assessed at the end of the quarter.

A couple of other commenters suggested changing the time periods for determining compliance (for both systems conducting quarterly monitoring and those conducting triennial monitoring). These recommendations included assessing compliance based on the results from eight consecutive quarterly samples (rather than four). For those systems conducting triennial monitoring, some commenters proposed that the compliance determination be based on one triennial sample result. For systems determining compliance through an RAA calculation, the EPA believes four consecutive quarterly samples is an adequate representation of the regulated PFAS levels while also assessing compliance in a timely manner. For systems conducting triennial monitoring, if a water system has a sample result at or above the EPA's trigger levels, the system will immediately be required to begin quarterly monitoring. This is consistent with other monitoring requirements for other SOCs and, given the change in measured concentration, will provide additional information over a consistent and longer period of time to better assess the average level of regulated PFAS within the water supply and ensure the water system is reliably and consistently below the MCL.

In the proposed rule, the EPA requested comment on whether the agency should consider an alternative to the approach of using zero when calculating the RAAs if a sample result is less than the PQL. Specifically, in the case where a regulated PFAS is detected but the result is below its proposed PQL, the proposed rule invited comment on whether the trigger level (proposed as one third of the PQL) should be used as the value in calculating the RAA for compliance purposes.

The EPA received numerous comments related to the proposed approach for calculating the RAA for compliance with the NPDWRs, particularly on the incorporation of sample results below the PQLs for the regulated PFAS (see sections V and VII for more information on PQLs.) Many commenters, including some states, supported the EPA's proposed approach to utilize zero for results below PQL to calculate the average for compliance purposes. These commenters cited the definition of the PQL as the lowest concentration of an analyte that can be reliably measured within specified limits of precision and accuracy during routine laboratory conditions and noted that this is a level that all laboratories should be able to achieve. Consequently, they suggested that values below these PQLs should not be used for the compliance calculation. Several of these commenters expressed concern that using estimated or other values with less precision in the compliance calculation could result in utilities needing to take actions to address levels of regulated PFAS that are not well-quantified and may not be representative of regulated PFAS levels. Many commenters suggested that since all laboratories cannot achieve values less than the PQLs, this would result in equity issues with respect to disparate laboratories capabilities. Some also suggested that the approach could exacerbate any potential laboratory capacity issues.

The EPA agrees with these commenters that values below the PQLs for the regulated PFAS should not be used in the compliance calculation. As cited previously by commenters and the EPA in sections V and VII, PQLs are the lowest concentration that can be reliably measured within specified limits of precision and accuracy during routine laboratory operations. As noted in the rule proposal, ''the agency must have a high degree of confidence in the quantified result as it may compel utilities to make potentially costly compliance decisions in order to comply with the MCL.'' Moreover, because compliance with the MCL is determined by analysis with approved analytical techniques, the ability to analyze consistently and accurately for a contaminant is important to enforce a regulatory standard. The EPA recognizes the potential for minor analytical variabilities within sampling procedures and laboratory analyses below the PQL and this approach offers operational certainty to utilities, provides assurances of precision and accuracy in the concentrations at or above the PQL that are achievable for all laboratories, ensures equitable access to all laboratories with comparable analytical capabilities for the purposes of compliance sample results, and reduces the potential for laboratory capacity issues.

Many other commenters did not support the EPA's proposed approach and offered that all sample results between method detection limits and PQLs, even if estimated, should be used. Alternatively, some suggested that any results that laboratories are able to quantify should be used in calculating the RAA for compliance. A subset of these commenters suggested that using zero (instead of an estimated or semi-quantitative value) biases the RAA compliance calculation, is even less precise and accurate than using the values below the PQLs, is contrary to the RAA compliance calculation for other SOC NPDWRs and demonstrates a reduction in public health protection. Some commenters also suggested that this could result in public communication challenges if laboratories are able to estimate or quantify values below the PQLs and zero is instead used in the calculation. Further, several commenters submitted that, in their experiences, some laboratories are capable of reliably and accurately reporting below the PQLs.

While the EPA recognizes that using zero for values below the PQL would result in a differing RAA compliance calculation result than if the values below the PQL were instead used, on a national scale, these values below the PQL do not consistently represent values with the precision, accuracy, and reliability the EPA believes are necessary for compliance determination purposes. Therefore, the EPA's national approach to achieve consistency (recognizing that laboratories have varying analytical capabilities) is to judge compliance based on results at or above the PQL. Using inconsistent values below the PQL may result in MCL compliance determination inequities across systems.

The EPA agrees that some laboratories are capable of reliably measuring the regulated PFAS below the EPA's PQLs. This is supported by a subset of state PFAS monitoring data that represents some sampling with quantified values below the EPA's PQLs. Further, in the March 2023 proposal, the EPA recognized that ''quantitation of the contaminants can be achieved between the method detection limit and the PQL'' though the EPA also noted in the proposal that this is ''not necessarily with the same precision and accuracy that is possible at and above the PQL.'' The EPA must set requirements evaluating the circumstances of all PWSs and laboratory capabilities throughout the country. The agency notes that states must establish requirements at least as stringent as the EPA to maintain primacy; however, under the Safe Drinking Water Act (SDWA), states with primacy may establish more stringent requirements. In instances where a laboratory can demonstrate it is capable of precisely and accurately quantifying values below the PQLs, some states may choose to establish their own requirements that are more stringent and use these values for the compliance calculation.

The agency also received a few comments on the possible alternative approach of using the proposed trigger level as the value in calculating the RAA for compliance purposes when the result is estimated as between the trigger level and PQL. Most commenters did not agree with using the trigger levels as an estimate instead of zero when values are below the PQL and noted that these values could result in inequitable implementation of the rule based on laboratory analytical capabilities.

After consideration of all these comments and for the reasons described previously, the EPA does not believe it is appropriate to use trigger level values or any other values above defined detection limits but below the PQL as part of the RAA compliance calculation based on the information available to the agency today. Trigger levels are appropriate to determine if the contaminant is present (*i.e.,* detected) and for the determination of reduced monitoring frequency, however the EPA concludes that values below the PQL would not consistently and reliably demonstrate the accuracy and reliability necessary for compliance determination purposes that can result in make potentially costly expenditures for PWSs.

3. Final Rule

For the final rule, the EPA is maintaining the proposed compliance calculation determination approach. For systems with sampling locations monitoring quarterly, compliance with the MCLs for regulated PFAS is determined by calculating RAAs using compliance results for particular sampling points. Based on final rule changes to the compliance monitoring requirements previously described in section VIII.A of this preamble above, systems with sampling locations monitoring less frequently than quarterly are required to revert to quarterly sampling for *all* regulated PFAS in the next quarter at each EP with the exceedance where either the sample result(s) are at or above the rule trigger level (for those on triennial monitoring) or the sample result(s) are at or exceed the MCL (for those on annual monitoring). In both cases, the triggered sample result is required to be used for the first quarter of monitoring in calculating the RAA. If a system takes more than one compliance sample during each quarter at a particular monitoring location, the system must average all samples taken in the quarter at that location to determine the quarterly average and this will be used in calculating the RAAs. Conversely, if a system does not collect the required compliance samples for a quarter, the RAA will be based on only those quarters where samples were collected during the past four quarters. A system will generally not be considered in violation of an MCL until it has completed one year of quarterly sampling (*i.e.,* a system on an annual or triennial monitoring schedule with an exceedance of the MCL is not in violation until it completes one year of quarterly sampling with the sample exceeding the MCL used as the sample result for the first quarter of the RAA). However, regardless of the result of subsequent monitoring, if a quarterly sample result will cause the RAA to exceed an MCL at any sampling point (*e.g.,* the first quarter sample result is greater than twice the MCL and the second quarter sample result is also greater than twice the MCL) or if an annual or triennial sample result causes the RAA to exceed an MCL at any sampling point (*i.e.,* the analytical result is greater than four times the MCL), then the system is out of compliance with the MCL immediately.

The EPA is also retaining the proposed approach for the MCL compliance calculation where, if a sample result is less than the PQL for the monitored PFAS, zero will be used to calculate the RAA (if monitoring quarterly). To clarify how to implement approach, the EPA is providing a few different examples related to calculating the RAA for the PFOA/PFOS MCLs, the individual MCLs for PFHxS, PFNA, and HFPO–DA, and the Hazard Index MCL for the mixtures of PFHxS, PFNA, HFPO–DA, and PFBS.

If a system conducting quarterly monitoring has sample results for PFOA that are 2.0, 1.5, 5.0, and 1.5 ng/L for their last four quarters at a sample location, the values used to calculate the RAA for that sample location would be 0, 0, 5.0, and 0 ng/L with a resulting PFOA RAA of 1.3 ng/L (*i.e.,* (0 + 0 + 5.0 + 0)/4 = 1.3 ng/L). For PFOA and PFOS, as described in section V of this preamble, the MCLs of 4.0 ng/L are promulgated with two significant figures and must be expressed as such in the calculation with any rounding not occurring until the end of the calculation. Data reported to the primacy agency must contain the same number of significant digits as the MCL. In calculating data for compliance purposes, the number must be rounded to two significant digits. The last significant digit should be increased by one unit if the digit dropped is 5, 6, 7, 8, or 9, and if the digit is 0, 1, 2, 3, or 4, the preceding number does not change (*e.g.,* 1.37 is reported as 1.4).

As described in section V of this preamble, the EPA is finalizing individual MCLs and Health Based Water Concentrations (HBWCs) for PFHxS (10 ng/L), HFPO–DA (10 ng/L), and PFNA (10 ng/L), the HBWC for PFBS (2000 ng/L), and the Hazard Index MCL (1 unitless) with one significant figure. Similar to PFOA and PFOS, if a sample result is less than the respective PQLs for these PFAS (*i.e.,* 3.0 ng/L for PFHxS, 5.0 ng/L for HFPO–DA, and 4.0 ng/L for PFNA), zero will be used to calculate compliance both for the PFHxS, PFNA, and HFPO–DA MCLs and the Hazard Index MCL for mixtures of PFHxS, PFNA, HFPO–DA, and PFBS. As an example, for the HFPO–DA MCL compliance calculation (which would be the same for the PFHxS and PFNA MCLs using their respective PQLs), if a system conducting quarterly monitoring has HFPO–DA sample results that are 3.2, 6.1, 5.5, and 2.7 ng/L for the last four quarters at a sample location, the values used to calculate the RAA for that sample location would be 0, 6.1, 5.5, and 0 ng/L with a resulting HFPO–DA RAA of 3 ng/L after rounding to one significant figure at the end of the calculation (*i.e.,* (0 + 6.1 + 5.5 + 0)/4 = 2.9 ng/L). Therefore, this system has not violated the MCL for HFPO–DA. The EPA notes that for all MCL RAA calculations, water systems are required to retain the unrounded RAA value (2.9 ng/L in this example) for use in the next RAA calculation as no rounding should occur until the end of the overall compliance calculation (*i.e.,* 2.9 ng/L, not 3 ng/L, should be used).

To provide an example calculation for determining compliance with the Hazard Index MCL for mixtures of PFHxS, PFNA, HFPO–DA, and PFBS, if the quarterly sample results at a sample location are 2.1 ng/L for PFHxS, 3.4 for HFPO–DA, 4.1 for PFNA, and 20.0 for PFBS, the water system would first determine the Hazard Index value for that quarter, which is 0.42 (*i.e.,* ((0/10) + (0/10) + (4.1/10) + (20.0/2000) = 0.42). To then calculate the RAA Hazard Index MCL, if the preceding three quarters had unrounded Hazard Index values of 0.76, 1.10, and 0.53 at the same sample location, the resulting RAA Hazard Index MCL would be 0.7 after rounding to one significant figure at the end of the calculation (*i.e.,* (0.76 + 1.10 + 0.53 + 0.42)/4 = 0.70). Consequently, this system has not violated the Hazard Index MCL.

C. Can systems use previously collected data to satisfy the initial monitoring requirement?

1. Proposal

The EPA proposed that systems be allowed to use previously collected monitoring data to satisfy the initial monitoring requirements. In general, a system with appropriate historical monitoring data for each EP, collected using EPA Methods 533 or 537.1 as part of UCMR 5 or a state-level or other appropriate monitoring campaign, could use that monitoring data to satisfy initial monitoring requirements. The EPA notes that for systems monitoring under UCMR 5, all surface water systems are required to collect four quarterly samples and all groundwater systems are required to collect two quarterly samples over a period of 12 months.

While the EPA expects most systems serving 3,300 or greater will have some UCMR 5 data, the EPA also proposed that systems with previously acquired monitoring data from outside UCMR 5, including state-led or other appropriate occurrence monitoring using EPA Methods 533 or 537.1 would also be permitted to use these other monitoring data in lieu of separate initial monitoring for regulated PFAS. The proposed approach may have allowed systems serving fewer than 3,300 (many of whom do not participate in UCMR 5) to otherwise satisfy the initial monitoring requirements. The EPA proposed that data collected after January 1, 2023, be accepted for EP samples, and data collected between January 1, 2019, and December 31, 2022, also be accepted if it is below the proposed rule trigger level of 1.3 ng/L for PFOA and PFOS and a Hazard Index of 0.33 for PFHxS, PFNA, HFPO–DA, and PFBS. Additionally, the EPA proposed that if systems have multiple years of data, the most recent data were to be used.

In the proposal, the EPA stated that if a system had conducted prior monitoring involving fewer than the number of samples required for initial monitoring under this PFAS NPDWR, then all surface water systems, GWUDI systems, and groundwater systems serving greater than 10,000 would be required to collect at least one sample in each quarter of a calendar year that was not acquired and groundwater systems serving 10,000 or fewer would be required to collect one sample in a different quarter of the calendar year than the one in which the previous sample was acquired.

2. Summary of Major Public Comments and EPA Responses

The EPA requested comment on the proposal to allow the use of previously acquired monitoring data to satisfy the initial monitoring requirements. This included a request for feedback on the data collection timeframe requirements and on whether particular QA requirements should be established for such data. Of commenters that provided input on the proposed allowance, nearly all supported the use of previously collected data to support the initial monitoring requirements. The EPA agrees with these commenters that appropriate, previously collected data should be allowed and notes that there will be significant data available from UCMR 5 monitoring and from the many states that have been proactively conducting PFAS drinking water monitoring. This will allow for a significant opportunity to reduce burden for numerous water systems, as well as decrease the potential for laboratory capacity issues. One commenter suggested that the use of this data may not be sufficiently representative of current PFAS concentrations in drinking water systems as the laboratory analyses previously used may not have been sufficiently sensitive to detect the analytes, relative to the proposed PFAS regulatory standards. The EPA disagrees with this commenter as the analytical methods proposed for PFAS analysis were available for the majority of the time period (*i.e.,* 2019 and after) in which data are allowed to be used to satisfy the initial monitoring requirements. Furthermore, the rule provides that a primacy agency may choose to not allow these data to satisfy initial monitoring requirements and may require more frequent monitoring on a system-specific basis. Additionally, the EPA clarifies that previous monitoring does not automatically qualify water systems for reduced compliance monitoring; rather it is the results from that monitoring that determine the eligibility for a reduced compliance monitoring schedule.

Many commenters suggested that the use of these data should be at the state's discretion and requested that the EPA provide additional flexibility to the primacy agencies in the determination of which data are allowed, including the number of samples and the QA requirements. Moreover, several commenters asked that the EPA clarify how much additional data would be needed to satisfy the initial monitoring requirements if a previous monitoring campaign included less sampling than required under the rule initial monitoring requirements. Specifically, a few commenters noted that, under the requirements of UCMR 5 monitoring, groundwater systems serving greater than 10,000 would have results from two sampling events, not the four needed to satisfy the initial monitoring requirements of this rule. Commenters requested that the EPA explain if these UCMR 5 systems would need to collect additional (supplemental) samples. A few commenters suggested UCMR 5 monitoring should sufficiently meet the requirements for all systems, even though the proposed rule requires quarterly sampling for all groundwater systems serving greater than 10,000.

Having considered the public comments, the EPA is establishing in the final rule that water systems that have collected fewer samples (under UCMR or other programs) than required in this rule for initial monitoring must conduct supplemental monitoring that allows them to meet the minimum requirements. Additional details on this requirement are in section VIII.C.3 of this preamble. In the case of UCMR 5, for example, groundwater systems serving greater than 10,000 will be required to collect two additional samples beyond the two collected for UCMR 5. For more information on the initial monitoring requirements, please see section VIII.A of this preamble.

Several commenters requested that the EPA clarify whether only samples collected under UCMR 5 would be allowed to fulfill initial monitoring requirements, or if data under other monitoring efforts, such as state monitoring, would also be acceptable. As provided in the proposal and final rule, a state may accept results from all appropriate monitoring efforts, as determined by the state, including, but not limited to, UCMR 5 and other state-led efforts.

Several commenters provided various comments related to QA requirements for previously collected data, including data analysis methods, minimum reporting levels, and data collection timeframe. A few commenters expressed that the EPA should allow the use of results from modified EPA methods and/or other state-developed analytical methods. The EPA disagrees with these commenters. While there are other methods that have been used for data collection and analysis, the EPA is requiring that any data used for this rule be collected and analyzed using Methods 533 and 537.1 to ensure consistency across analytical results, as well as to align with the final rule analytical method requirements described in § 141.901. A few commenters requested that the EPA provide additional information on reporting level requirements of the data, with one commenter suggesting that the EPA should not allow this data to be used for initial monitoring purposes if the reporting limits of the laboratory are higher than the EPA's proposed PQLs. The rule provides that the available data can be used regardless of reporting or detection limits to satisfy the initial monitoring requirements; however, given these factors, the results may not support determinations for reduced compliance monitoring. Regarding data collection timeframes, a few commenters questioned why data collected prior to 2023 would not be accepted where the results are higher than the proposed rule trigger levels. In response, the EPA has modified the rule to allow data from January 1, 2019, and later to satisfy initial monitoring requirements, even if it is not below the final rule trigger levels if it meets all other requirements (including being analyzed using Methods 533 and 537.1). Data collected prior to 2019 may not be representative of water quality conditions and likely would not have been analyzed using these methods (given when they were published). The EPA notes if the results exceed the final rule trigger levels the system will not be eligible for a reduced monitoring schedule at that EP.

3. Final Rule

The EPA is retaining the proposed allowance of using previously collected monitoring data to satisfy some or all of the initial monitoring requirements. The agency notes that while use of this data is allowed, water systems may choose to conduct additional monitoring to satisfy their initial monitoring requirement in lieu of using pre-existing data. As described previously in section VIII.A of this preamble, the final rule initial monitoring requirements specify that all system sizes with surface water or GWUDI sources and groundwater systems serving greater than 10,000 are required to collect four quarterly samples, and groundwater systems serving 10,000 or fewer are required to collect two samples. The EPA is clarifying that the number of samples required is based at the EP; therefore, if a system serving 10,000 or fewer has EP with different source water types, the required monitoring is based on the source water type of that EP (*i.e.,* a system serving 10,000 or fewer that has surface water, groundwater, and/or GWUDI sources during the initial monitoring period must collect two samples at the EP sourced by groundwater and four samples at the EP sourced by surface water or GWUDI). For systems serving 10,000 or fewer that change the source water type at EP throughout the initial monitoring period (*i.e.,* one part of the year is surface water, and the remaining part of the year is groundwater and/or GWUDI), the EP must follow the sampling requirements of surface water systems.

In the final rule under § 141.902(b)(1)(viii), the EPA is maintaining that if a system has some previously collected results, but fewer than the number required to satisfy the initial monitoring requirements, they must conduct additional monitoring such that it, coupled with the previous monitoring, meets the requirements of this rule. All surface water and GWUDI systems, and groundwater systems serving greater than 10,000, must collect the required additional samples 2–4 months apart from the months with available data, without regard to year, such that all quarters are represented (see section VIII.A of this preamble for more information).

In § 141.902(b)(1)(vi), the final rule maintains the requirement that the data must have been collected and analyzed using EPA Methods 533 or 537.1, and eliminates the requirement that data collected between January 1, 2019, and December 31, 2022, must reflect the laboratory's ability to measure at or below the rule trigger level to satisfy initial monitoring requirements. Data collected before January 1, 2019, cannot be used to satisfy these requirements. Additionally, any results above the final rule trigger levels of 2.0 ng/L each for PFOA and PFOS, 5 ng/L each for PFHxS, PFNA, and HFPO–DA, and a Hazard Index of 0.5 for PFHxS, PFNA, HFPO–DA, and PFBS would not allow the associated EP to be eligible for reduced monitoring.

D. Can systems composite samples?

1. Proposal

Subpart C of 40 CFR 141.24 describes instances where primacy agencies may reduce the samples a system must analyze by allowing samples to be composited. Composite sampling can potentially reduce analytical costs because the number of required analyses is reduced by combining multiple samples into one and analyzing the composited sample. However, in the proposal, the EPA noted that based on input the agency received from consulting with state regulators and small business entities (operators of small PWSs), PFAS are ubiquitous in the environment at low concentrations, which necessitates robust laboratory analytical precision at these low concentrations. Based on these potential implementation issues, the EPA proposed that compositing of samples would not be allowed.

2. Summary of Major Public Comments and EPA Responses

The EPA received comments related to composite sampling. The majority of these commenters agreed with the EPA's proposal to not allow samples to be composited due to analytical limitations and the increased potential for background contamination, along with the physical and chemical characteristics of PFAS. A few commenters suggested that they believed composite sampling could be implemented and would reduce the cost of analyses. Further, some of these commenters suggested that with proper guidelines and procedures for analyzing samples, possible contamination issues could be mitigated and asserted that issues with false negative and positive samples also impact discrete samples (*i.e.,* that they are not unique to composite sampling).

The EPA received other comments regarding the specifics of composite monitoring. One commenter noted grab samples as more appropriate and suggested that individual systems be permitted to request alternative sampling methodologies if needed. One other commenter suggested that compositing samples from varying EP should not be allowed. In addition, one commenter requested that the EPA provide information as to the increased risk of compositing samples, along with discussion of the proposed departure from the SMF for SOC ahead of rule finalization.

For commenters who offered that composite sampling could be implemented, the EPA agrees it would potentially decrease sampling analysis costs and that sampling errors can occur when handling and analyzing discrete samples. However, the compositing of samples necessarily involves additional handling, opening, and transfer steps than are required for the collection and analysis of individual samples. Therefore, the combining of samples that must be done for composite sample analysis represents an increased risk of sampling error, which could result in decreased public health protection and additional sampling costs. The agency also does not agree that alternative sampling methodologies should be permitted and requires the use of EPA Methods 533 and 537.1 for monitoring per the requirements of the rule. Please see section VII of this preamble for more information on methods.

As discussed previously, PFAS are pervasive in the environment and require robust laboratory analytical precision, particularly at low concentrations. Accordingly, the EPA agrees with commenters that do not support the allowance of composite sampling and maintains that discrete sampling is the most appropriate type of sampling for regulated PFAS.

3. Final Rule

Based on consideration of public comments (many of which supported the EPA's concerns about the ubiquitous nature of PFAS at low concentrations in the environment, the necessary robust laboratory analytical precision required, and potential implications for implementation), the final rule does not allow composite samples.

E. Can primacy agencies grant monitoring waivers?

1. Proposal

Subpart C of 40 CFR 141.24 describes instances where the primacy agency may grant waivers predicated on proximity of the system to contaminant sources (*i.e.,* susceptibility to contamination) and previous uses of the contaminant within the watershed (including transport, storage, or disposal). The EPA did not include a provision to allow primacy agencies to grant monitoring waivers as a regulatory flexibility in the proposed rule. The EPA did, however, request public comment on whether to allow systems to apply to the primacy agency for a monitoring waiver of up to nine years (one full compliance cycle) if, after at least one year of quarterly sampling, the results are below the rule trigger level, or for systems that may be approved for reduced monitoring, if at least two consecutive results are below the rule trigger level. The EPA also requested comment on allowing similar monitoring waivers to be granted based on previously acquired monitoring data as described in section VIII.C of the preamble for the proposed rulemaking. The EPA additionally sought comment on possible alternatives to traditional vulnerability assessments that should be considered in order to identify systems as low risk and potentially eligible for monitoring waivers.

2. Summary of Major Public Comments and EPA Responses

Several commenters suggested that monitoring waivers should not be allowed for this rule. Several additional commenters cited the persistence and mobility of PFAS in the environment and advised that reduced monitoring frequencies should be no less than every three years on the basis that drinking water consumers in unmonitored areas may unknowingly be exposed to these PFAS. Furthermore, many other commenters suggested that PFAS contamination can migrate significantly over a three-year period.

Many other commenters were supportive of monitoring waivers for this rule under certain circumstances. Several commenters indicated that waivers would be appropriate if they were based on monitoring results. A few commenters recommended that if monitoring waivers were to be allowed, that they should not be based solely on a traditional vulnerability assessment. A couple of commenters stated that waivers based on vulnerability alone should not be allowed during the initial monitoring period. One commenter recommended waiting until UCMR 5 monitoring is complete before allowing monitoring waivers to be granted through vulnerability assessments. A couple of commenters suggested that waivers be considered if they are based on a combination of vulnerability *and* monitoring results, while one other commenter suggested that assessing watershed characteristics to demonstrate eligibility for monitoring waivers would be protective of chronic health risks. One commenter noted that merely allowing waivers to be granted would not necessarily reduce public health protection under the rule, as primacy agencies will retain the ability to deny waiver applications.

After consideration of these comments, and due to the mobility and persistence characteristics of the regulated PFAS, the final rule does not allow monitoring waivers. These specific properties of the regulated PFAS and their observed ubiquity in both drinking water and within many other sources make waivers impractical and complicate the ability to maintain public health protection if such a provision were included as part of this rule. Moreover, the EPA is not confident that allowing monitoring any less frequently than every three years or conducting vulnerability assessments will accurately capture potential concentration variations over the long term or protect against risks from new contamination sources.

3. Final Rule

Consistent with the proposal, the final rule does not include a provision that would allow primacy agencies to issue monitoring waivers. These waivers would increase the potential for public health risks and the EPA does not consider them necessary to reduce burdens on primacy agencies, water systems and communities given the other flexibilities provided in the rule.

F. When must systems complete initial monitoring?

1. Proposal

Pursuant to section 1412(b)(10), the proposed rule required compliance with all aspects of the NPDWR three years after promulgation. This included satisfying initial monitoring requirements as described in sections VIII.A and VIII.C within the three years following rule promulgation.

2. Summary of Major Public Comments and EPA Responses

In the proposal, the EPA requested public comment on the proposed initial monitoring timeframe, particularly for NTNCWS or all systems serving 3,300 or fewer. Many commenters expressed support for the EPA requiring initial monitoring as soon as possible with a few commenters explicitly supporting the EPA's proposed initial monitoring timeframe noting it allows sufficient time for water systems to comply with the initial monitoring requirements. However, other commenters suggested that water systems would not be able to utilize the full three years following rule promulgation to perform initial monitoring and take actions to ensure compliance with the MCL if monitoring results showed elevated levels of PFAS. While the agency agrees that it may be difficult to conduct initial monitoring and take necessary remedial actions (*e.g.,* treatment installation) within three years, the EPA finds that it is practicable for all systems to complete their initial monitoring within three years. This is particularly the case since the large majority of systems serving greater than 3,300 will have sufficient monitoring data from UCMR 5 and many other systems will have at least some data to satisfy the rule's initial monitoring requirements. Moreover, as described in section XI.D. of this preamble, the EPA is exercising its authority under SDWA section 1412(b)(10) to implement a nationwide two-year capital improvement extension to comply with MCL. Consequently, water systems will have up to the full three years following rule promulgation to plan and conduct monitoring and still have two additional years to complete any actions needed to comply with the MCLs.

Several commenters suggested that the EPA consider a staggered initial monitoring timeframe by system size, such as those used in other previous NPDWRs, where, for example, larger sized systems conduct monitoring first followed by smaller systems. In the examples provided by commenters, this staggered monitoring could also allow systems to achieve compliance on a staggered schedule. A few commenters suggested that this is necessary to address potential laboratory capacity issues and to allow smaller systems additional time to plan and obtain resources to conduct the monitoring. The EPA disagrees that staggering the monitoring requirements to allow different compliance dates is necessary. SDWA 1412(b)(10) specifies that all systems must demonstrate compliance three years following rule promulgation except where a state or the EPA may grant an extension of up to two additional years to comply with MCL(s) if the EPA or the state (for an individual system) needs additional time for capital improvements. Therefore, the intent of the statute is to allow extensions to complete the capital improvements necessary to comply with the MCL. The EPA considers the three years sufficient for completing the rule's initial monitoring requirement. The EPA's allowance of previously collected monitoring data will also significantly reduce the potential for laboratory capacity challenges. As previously noted in section VIII.A of this preamble, the EPA has revised the required intervals between samples collected for initial monitoring under this rule to closely parallel the intervals required for UCMR 5, to promote the useability of existing data.

The EPA is not prescribing any staggering of monitoring (*e.g.,* based on system size) but encourages primacy agencies to work with the systems they oversee to ensure their initial monitoring occurs and adjust schedules (within the three years following rule promulgation) as appropriate.

3. Final Rule

The EPA is finalizing the requirement that initial monitoring, or demonstration of previously collected data to satisfy initial monitoring requirements, must be completed within the three years following rule promulgation (*i.e.,* April 26, 2027) to ensure that water systems have the information needed to inform decisions to meet the MCL compliance date. As described previously and in section XI.D, the EPA is providing a two-year capital improvement extension under SDWA 1412(b)(10), allowing additional time for those systems to comply with the MCL. Requiring water systems to conduct initial monitoring within the three years following rule promulgation will ensure public health protection as soon as practicable and allow these water systems to maximize utilization of the capital-improvement extension time. Additionally, the flexibility in the final rule for systems to use previously acquired monitoring data to satisfy some or all of their initial monitoring will reduce the potential for laboratory capacity challenges. The EPA encourages systems that may not have available data and/or choose to conduct additional monitoring to conduct their initial monitoring as soon as practicable following rule promulgation to allow for remedial actions that may needed, based on monitoring results, and to comply with the MCL by the compliance date.

G. What are the laboratory certification requirements?

1. Proposal

The EPA proposed that laboratories demonstrate their capability to meet the objectives of this regulation. The proposal would require laboratories to analyze performance evaluation (PE) samples every year for each method and contaminant in order to achieve and maintain certification from their primacy agency.

2. Summary of Major Public Comments and EPA Responses

A few commenters requested that the EPA develop guidance and training for drinking water laboratory certification programs to evaluate laboratories seeking certification. The EPA agrees that training for laboratory certification officers is appropriate. The EPA will develop training materials and guidance for drinking water certification programs to evaluate laboratories to ensure adherence to the requirements of EPA Methods 533 and 537.1 (USEPA, 2005b).

One commenter requested that the EPA establish reciprocity between laboratory certification programs to utilize all potential laboratory capacity available. As described in the EPA's *Manual for the Certification of Laboratories Analyzing Drinking Water,* laboratory certification programs may recognize drinking water laboratory certifications (or comparable ''accreditation'') from other laboratory certification programs, by reciprocity (USEPA, 2005b). Most laboratory certification programs do utilize the practice of reciprocal certification. Reciprocal certification can only be granted to laboratories utilizing EPA Methods 533 and 537.1.

3. Final Rule

Under the final rule, certified laboratories must demonstrate their capability to meet the objectives of this regulation. Laboratories are required to analyze PE samples every year for each method and contaminant in order to achieve and maintain certification from their primacy agency.

H. Laboratory Quality Assurance/Quality Control

In the proposal, the EPA requested comment on other monitoring-related considerations including quality assurance/quality control (QA/QC) associated with drinking water sampling and analysis.

Many commenters suggested the potential for false positives to misrepresent actual levels of the regulated PFAS within the drinking water sample due to the ubiquity of PFAS and the possible background interference. The EPA is aware of the potential for background contamination due to the ubiquitous nature of PFAS in the environment. The EPA agrees that PFAS sampling is highly sensitive and there is potential for sample contamination. However, with proper training tools and communications, that potential can be mitigated, though not sufficiently enough to allow for composite sampling as discussed in section VIII.D of this preamble. For example, the UCMR program has released several sampling guidance documents and a small-systems sampling video to assist small and medium utilities with the PFAS sampling. These products have also been distributed to the UCMR laboratory community, which has been encouraged to share them with their PWS clients.

Also, Method 533 and Method 537.1 require the analysis of an LRB with each extraction batch. If method analytes are detected at or above ⅓ the minimum reporting level, suggestive of background contamination, all positive field sample results associated with that extraction batch are invalid for the impacted analytes. Both methods also require the analysis of an FRB (a blank that is prepared at the sampling location) when any PFAS are detected above the minimum reporting level in field samples. The use of laboratory and field blanks were incorporated into the methods as QC to reduce the potential for false positives due to background contamination.

IX. Safe Drinking Water Act (SDWA) Right To Know Requirements

A. What are the Consumer Confidence Report requirements?

1. Proposal

A community water system (CWS) must prepare and deliver to its customers an annual Consumer Confidence Report (CCR) in accordance with requirements in 40 CFR part 141, subpart O. A CCR provides customers with information about their local drinking water quality as well as information regarding the water system's compliance with drinking water regulations. The EPA proposed that CWSs be required to report detected PFAS in their CCRs, specifically, PFOA, PFOS, PFHxS, PFNA, HFPO–DA, and PFBS, and the Hazard Index for mixtures of PFHxS, PFNA, HFPO–DA, and PFBS. The EPA also proposed adding paragraph (g) to 40 CFR 141.154 that would require health effects language be provided when any regulated PFAS is measured above the maximum contaminant level (MCL), in addition to those with an MCL violation.

2. Summary of Major Public Comments and EPA Responses

A few commenters requested clarification of the health effects language included in the CCR. Specifically, a couple of commenters said the proposed standard health effects language included in the CCR for a Hazard Index MCL exceedance was not clear. Commenters found some of the language regarding the Hazard Index MCL to be confusing and offered suggestions for clarification. The EPA has considered this input and revised the health effects language associated with PFAS exposure, including the Hazard Index.

A few of commentors raised concerns about requiring reporting of results below the practical quantitation level (PQL) in the CCR as these data may not be quantified with what they deem is appropriate precision. One commentor requested that any detected PFAS, not just the six regulated contaminants, be reported in the CCR. The EPA disagrees with commenters who voice concern over reporting measurements below the PQLs for PFOA and PFOS as ''detected'' contaminants in the CCR. Reporting these measurements in the CCR will allow customers to understand that the contaminant was detected in the water supply. While measurements below the PQL will not be used to calculate compliance with MCLs for the final rule, measurements lower than the PQL are achievable by individual laboratories, and therefore these measurements can be used for screening, to determine compliance monitoring frequency, and to educate consumers about the existence of PFAS (for further discussion of PQLs for regulated PFAS, please see section VII of this preamble). As such, the EPA believes that measurements below the PQL can reasonably be reported as ''detected'' for purposes of the CCR. This requirement is consistent with the CCR Rule in 40 CFR 141.153(d) which requires CWSs to report information on detected contaminants for which monitoring was required by the EPA or the state. The CCR reporting requirement includes unregulated contaminants for which monitoring is required pursuant to the Unregulated Contaminant Monitoring Rule (UCMR) as well as regulated contaminants in accordance with SDWA (Safe Drinking Water Act) 1414(c)(4). If the system has performed additional monitoring, the EPA strongly encourages them to include the results in the CCR, consistent with 40 CFR 141.153(e)(3).

3. Final Rule

As part of this action, the EPA has modified the trigger level value for quarterly monitoring from one-third of the MCL to one-half of the MCL in response to concerns that laboratories would not have the capacity to consistently measure as low as the threshold of one-third of the MCL (for further discussion of the EPA's trigger levels for the final rule, please see section VIII of this preamble). To reflect this change in the trigger level, the EPA has modified 40 CFR 141.151(d), which identifies what is considered detected for purposes of reporting in CCRs consistent with SDWA 1414(c)(4). The EPA had also proposed adding a provision to require CWSs that detect any PFAS above the MCL to include health effects language for PFAS and stated in the preamble for the rule proposal that CWSs would be required to report detected PFAS as part of their CCRs. Because SDWA 1414(c)(4)(B) specifies that the Administrator may only require health effects language be reported in the CCR for situations other than an MCL violation for not more than three regulated contaminants, the EPA has removed the amendment to paragraph (g) of 40 CFR 141.154 included in the proposed rule from the final rule and has instead updated appendix O to part 141 for the final rule to only require CWSs that have violations of the PFAS MCLs to include health effects language for PFAS. Since systems must complete initial monitoring within three years of rule promulgation, systems will be required to report results and other required information in CCRs beginning with 2027 reports. As the MCL compliance date is set at five years following rule promulgation, systems will be required to report MCL violations in the CCR, accompanied by the required health effects language and information about violations, starting in 2029.

The EPA acknowledges the need to protect public health with clear and concise language that outlines the risks associated with exposures exceeding the MCLs and Hazard Index. The EPA's broad review of the most current research provides a comprehensive understanding of how exposure to PFAS may result in adverse impacts on the health of individuals. In response to commenter requests for plain language explanations of the Hazard Index, the EPA is adding the following definition of the *Hazard Index* in 40 CFR 141.153(c)(3)(v) of the CCR Rule to improve clarity and understandability for consumers (for more information on how the Hazard Index is calculated for this rule, please see table to paragraph (b) under 40 CFR 141.50):

Hazard Index or HI: The Hazard Index is an approach that determines the health concerns associated with mixtures of certain PFAS in finished drinking water. Low levels of multiple PFAS that individually would not likely result in adverse health effects may pose health concerns when combined in a mixture. The Hazard Index MCL represents the maximum level for mixtures of PFHxS, PFNA, HFPO–DA, and/or PFBS allowed in water delivered by a public water system. A Hazard Index greater than one (1) requires a system to take action.

Additionally, in response to commenters' request for clearer mandatory health effects language, the final rule includes revised mandatory health effects language required as part of CCRs, in cases when MCL violations have occurred.[10] Identical mandatory health effects language is also required for public notification (PN) under the final rule (PN requirements are described further in section IX.B of this preamble). The mandatory health effects language in the final rule reads as follows:

Health effects language for PFOA: Some people who drink water containing PFOA in excess of the MCL over many years may have increased health risks such as cardiovascular, immune, and liver effects, as well as increased incidence of certain types of cancers including kidney and testicular cancer. In addition, there may be increased risks of developmental and immune effects for people who drink water containing PFOA in excess of the MCL following repeated exposure during pregnancy and/or childhood.

Health effects language for PFOS: Some people who drink water containing PFOS in excess of the MCL over many years may have increased health risks such as cardiovascular, immune, and liver effects, as well as increased incidence of certain types of cancers including liver cancer. In addition, there may be increased risks of developmental and immune effects for people who drink water containing PFOS in excess of the MCL following repeated exposure during pregnancy and/or childhood.

Health effects language for PFHxS: Some people who drink water containing PFHxS in excess of the MCL over many years may have increased health risks such as immune, thyroid, and liver effects. In addition, there may be increased risks of developmental effects for people who drink water containing PFHxS in excess of the MCL following repeated exposure during pregnancy and/or childhood.

Health effects language for PFNA: Some people who drink water containing PFNA in excess of the MCL over many years may have increased health risks such as elevated cholesterol levels, immune effects, and liver effects. In addition, there may be increased risks of developmental effects for people who drink water containing PFNA in excess of the MCL following repeated exposure during pregnancy and/or childhood.

Health effects language for HFPO–DA: Some people who drink water containing HFPO–DA in excess of the MCL over many years may have increased health risks such as immune, liver, and kidney effects. There is also a potential concern for cancer associated with HFPO–DA exposure. In addition, there may be increased risks of developmental effects for people who drink water containing HFPO–DA in excess of the MCL following repeated exposure during pregnancy and/or childhood.

Health effects language for Hazard Index PFAS: Per- and polyfluoroalkyl substances (PFAS) can persist in the human body and exposure may lead to increased risk of adverse health effects. Low levels of multiple PFAS that individually would not likely result in increased risk of adverse health effects may result in adverse health effects when combined in a mixture. Some people who consume drinking water containing mixtures of PFAS in excess of the Hazard Index (HI) MCL may have increased health risks such as liver, immune, and thyroid effects following exposure over many years and developmental and thyroid effects following repeated exposure during pregnancy and/or childhood.

B. What are the Public Notification (PN) requirements?

1. Proposal

As part of SDWA, the PN Rule ensures that consumers will know if there is a problem with their drinking water. Notices alert consumers if there is risk to public health. They also notify customers: if the water does not meet drinking water standards; if the water system fails to test its water; if the system has been granted a variance; or if the system has been granted an exemption (that is, more time to comply with a new regulation).

All public water systems (PWSs) must give the public notice for all violations of National Primary Drinking Water Regulations (NPDWRs) and for other situations. Under the EPA's PN Rule, the public notice requirements for each violation or situation are determined by the tier to which it is assigned. The EPA specifies three categories, or tiers, of PN requirements, to take into account the seriousness of the violation or situation and any potential adverse health effects that may occur (USEPA, 2000f). The

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[10] The EPA has developed the existing mandatory health effects language to communicate accurate, clear health information to a non-technical audience. Although the EPA believes additional detail is not necessary to include in the mandatory health effects language which is required only where MCL violations have occurred, the EPA also recognizes that, in general, a single exposure at a critical time in development may produce an adverse developmental effect (see USEPA, 1991a).

EPA proposed that violations of the three MCLs in the proposal be designated as Tier 2 and as such, PWSs would be required to comply with 40 CFR 141.203. Per 40 CFR 141.203(b)(1), notification of an MCL violation should be provided as soon as practicable but no later than 30 days after the system learns of the violation. The proposed rule also designated monitoring and testing procedure violations as Tier 3, which would require systems to provide notice no later than one year after the system learns of the violation. The system would then be required to repeat the notice annually for as long as the violation persists.

2. Summary of Major Public Comments and EPA Responses

Many commenters support the Tier 2 PN requirement for MCL violations. Commenters assert that Tier 2 notification is appropriate and consistent with other MCLs for chemicals with chronic effects. Conversely, many commenters suggest that the PN tiering be raised from Tier 2 to Tier 1 or that the EPA consider other PN approaches given concerns about health impacts resulting from exposure on timescales shorter than chronic exposure. Commenters assert that raising PN for MCL violations from Tier 2 to Tier 1 would ensure that consumers are informed of potential harm associated with elevated PFAS levels in a timelier manner so they can make informed risk management decisions. Additionally, a few commenters request the EPA re-categorize repeat MCL violations to Tier 3 due to the expected length of time needed for a PWS to design and construct treatment. Commenters argue that quarterly PN would not offer added value and could possibly result in confusion for consumers.

The EPA agrees with commenters that Tier 2 PN is appropriate for MCL violations based on analysis of a wide range of scientific studies that shows that long-term exposure may have adverse health effects. The EPA disagrees with commenters who recommend issuing Tier 1 notification for MCL violations. Tier 1 notices must ''be distributed as soon as practicable, but no later than 24 hours, after the public water system learns of the violation'' pursuant to section 1414(c)(2)(C)(i) of SDWA. The PN Rule preamble characterizes contaminants with violations routinely requiring a Tier 1 notice as those with ''a significant potential for serious adverse health effects from short-term exposure'', stating that other violations do not require Tier 1 notice because elevated levels of these contaminants are not ''strongly or consistently linked to the occurrence of the possible acute health effects'' (USEPA, 2000f). The EPA has not characterized health risks resulting from acute exposure (*i.e.,* < or = 24 hours) to PFAS and the EPA believes that issuing Tier 2 PN for MCL violations constitutes a health protective approach given that the MCLG values are based on health effects that occur after chronic exposure to PFAS (*i.e.,* cancer). Based on the available health effects information, the EPA has characterized developmental effects, including immune impacts, associated with developmental PFAS exposure in addition to health effects that occur after chronic exposure. The agency considers it reasonable to notify consumers within 30 days of a PWS learning of an MCL violation because it generally provides protection of the adverse health effects that may occur from exposure to PFAS during sensitive lifestages such as gestation. The EPA typically reserves Tier 1 notifications for acutely toxic contaminants. For example, nitrate, nitrite, or total nitrate and nitrite require Tier 1 notice because exceedances can result in immediate life-threatening health impacts for infants (*i.e.,* methemoglobinemia). Based on the currently available information, the developmental and chronic effects associated with exposure to these PFAS are not known to represent immediate acute health effects. For more information on the EPA's characterization of health effects resulting from PFAS exposure, please see (USEPA, 2024c; USEPA, 2024d). This approach is also consistent with the PN requirements for other synthetic organic contaminants regulated under SDWA. The EPA acknowledges that there may be instances in which it is appropriate to elevate the tiering of PN on a case-by-case basis. Under the existing PN Rule in 40 CFR 141.202(a), a violation that routinely requires a Tier 2 notice but poses elevated risk from short-term exposure may be elevated to Tier 1 at the discretion of the primacy agency (USEPA, 2000f). Additionally, the EPA will develop appropriate implementation guidance to assist in the understanding of PN requirements among other final rule requirements.

The EPA disagrees with commenters that recommended reclassifying ongoing MCL violations to Tier 3 for repeat notices. The EPA believes there is sufficient flexibility in the existing PN Rule 40 CFR 141.203(b)(2) that allows primacy agencies to allow a less frequent repeat notice on a case-by-case basis for unresolved violations, but no less than once per year, and the determination must be in writing. The EPA believes repeat notices are valuable to consumers that may not receive the initial notice and allow water systems to provide any updates to consumers, such as actions being taken to resolve the situation and estimated timelines.

A few commenters recommended that the EPA update the proposed PN health effects language. Commenters stated that the proposed health effects language was confusing and needed to be clarified as it did not sufficiently explain the health effects resulting from PFAS exposure. Additionally, commenters stated that further clarifying the health effects language would mitigate confusion from customers when receiving PN from their water system.

The EPA agrees with commenters that additional explanation of the health effects of PFAS exposure will more effectively communicate risk to consumers when they receive PN from their water system. The EPA has considered this input and has revised health effects language for the final rule to further clarify the health effects associated with PFAS exposure.

3. Final Rule

The final rule requires the PN of violations of all MCLs promulgated under this final rule to be designated as Tier 2 and as such, PWSs would be required to comply with 40 CFR 141.203. The final rule also designates monitoring and testing procedure violations as Tier 3, requiring systems to provide notice no later than one year after the system learns of the violation. Systems are also required to repeat the notice annually for as long as the violation persists. As systems must comply with initial monitoring requirements within three years of rule promulgation, systems will be required to provide Tier 3 notification for monitoring and testing procedure violations starting in 2027. As the MCL compliance date is set at five years following rule promulgation, systems will be required to provide Tier 2 notification for MCL violations starting in 2029. However, the EPA acknowledges that primacy agencies have the authority in the existing PN Rule (table 1 to § 141.201) to require systems to provide notices to consumers prior to the MCL compliance date. The EPA encourages primacy agencies to use this flexibility to require systems to provide notices to consumers for PFAS detections that precede the date that MCL compliance will take effect, as they deem appropriate. By encouraging systems to provide timely notification, it allows customers to take actions to protect their health, such as using a filter, while systems take necessary steps to apply treatment.

With respect to violations and reporting associated with the individual MCLs and Hazard Index MCL, the EPA recognizes that a utility may have two or more of these PFAS present that, over the course of four quarterly samples, may result in violation of multiple MCLs. For example, if, following four quarterly samples, a utility has PFHxS and HFPO–DA present and the RAA is above their respective MCLs and HBWCs of 10 ng/L, the system would be in violation of both the individual MCLs for PFHxS and HFPO–DA, as well as the Hazard Index MCL. Issuing multiple notifications (three in this example) for these violations may cause public confusion as the adverse health effects and exposure concern in this instance is not meaningfully different from either a Hazard Index or individual MCL perspective. To simplify implementation of PN in this scenario, the EPA is finalizing requirements in appendix A to subpart Q of part 141 such that utilities who violate the Hazard Index MCL and one or more individual MCLs because of the same compounds can issue one notification to satisfy the PN requirements for the multiple violations.

The EPA has also made edits to clarify the mandatory health effects language required in the PN of an MCL violation, as well as the CCR. The mandatory health effects language required for both PN and CCRs is summarized in section IX.A.3 of this preamble above.

X. Treatment Technologies

Section 1412(b)(4)(E) of the Safe Drinking Water Act (SDWA) requires that the agency ‘‘list the technology, treatment techniques, and other means which the Administrator finds to be feasible for purposes of meeting [the MCL],’’ which are referred to as best available technologies (BATs). The EPA generally uses the following criteria for identifying ‘‘feasible’’ BATs: (1) The capability of a high removal efficiency; (2) a history of full-scale operation; (3) general geographic applicability; (4) reasonable cost based on large metropolitan water systems; (5) reasonable service life; (6) compatibility with other water treatment processes; and (7) the ability to bring all the water in a system into compliance. Section 1412(b)(4)(E)(ii) of SDWA requires that the agency identify small system compliance technologies (SSCTs), which are affordable treatment technologies, or other means that can achieve compliance with the maximum contaminant level (MCL).

In the proposed rule, the EPA requested comments on: technologies designated as BATs, costs associated with nontreatment options; whether employing these treatment technologies are sound strategies to address PFAS as well as whether the BATs could feasibly treat to below the proposed MCLs; the type of assistance that would help public water systems (PWSs); potential benefits from co-removal; treatment residual disposal estimates; the capacity to address the increased demand for BATs as well as residuals disposal or reuse; impacts that PFAS residuals disposal may have in communities adjacent to the disposal facilities; the most appropriate disposal means for PFAS contaminated residuals and waste the systems may be generating; and SSCT selection as well as national affordability analysis, specifically on the methodologies.

A. What are the best available technologies?

1. Proposal

The agency proposed GAC, AIX, NF, and RO as BATs for the six PFAS under consideration in the proposed rule. The EPA also acknowledged that there are nontreatment options which may be used for compliance such as replacing a PFAS-contaminated drinking water source with a new uncontaminated source. The EPA also stated that conventional and most advanced water treatment methods are ineffective at removing PFAS.

2. Summary of Major Public Comments and EPA Responses

The vast majority of comments germane to the BAT designations support the EPA's designation of granular activated carbon (GAC), anion exchange resins (AIX), and high-pressure membranes (nanofiltration (NF) and reverse osmosis (RO)) as BATs that are technologically feasible for treating drinking water to the proposed standards or below. Many commenters shared practical experience with installed treatment including successes, costs, implementation considerations, challenges, and other areas. The EPA agrees that GAC, AIX, RO, and NF are BATs and consistent with the criteria outlined in the BAT/SSCT document for identifying ‘‘feasible’’ treatment for PFAS in this rule, and the comments providing information on practical full-scale experience with these technologies further support for this finding.

A few commenters suggested either that the designated BATs could not treat to or below the MCL or that not enough data was available to support the conclusion that the BATs could treat to at or below the proposed MCL. The EPA disagrees with these commenters based on the history of full-scale use as documented in the *Best Available Technologies and Small System Compliance for Per- and Polyfluoroalkyl Substances (PFAS) in Drinking Water* document (USEPA, 2024l), the information in the rule preamble, and in the comments that provided full-scale data as well as case studies. For example, commenters highlighted more than 45 military installations that have treated PFAS, including those in this rule, successfully for more than 15 years, a major water treatment company provided information on over 150 successful installations they had performed, and comments supported that there are significant numbers of industrial users successfully treating PFAS, including those in this rule. One commenter noted the example of the Chemours Fayetteville facility which used GAC to eliminate PFAS, including those in this rule, as high as 345,000 ng/L and has reduced PFAS in effluent to non-detect levels for several PFAS. Finally, the Water Quality Association reviewed proprietary performance data from its accredited laboratory demonstrating that this standard is feasible for the BATs selected to effectively remove the PFAS regulated in this rule from drinking water.

Many commenters pointed out site-specific issues with particular BATs. The EPA acknowledges that not every BAT represents the best treatment option for an individual system and site-specific considerations can limit BAT selection. For instance, residuals management considerations can limit the choice of RO/NF; particularly in states with limited water resources. While many commenters agreed that high pressure membranes such as RO and NF can remove the six PFAS included in the proposal, many commenters also suggested that high pressure membranes may not be the most feasible treatment option for some systems because of residual management considerations, which are discussed in the residuals management section. There are, however, documented RO/NF facilities for treating PFAS in California, Illinois, North Carolina, and Alabama (USEPA, 2024l). In response to public comment and residual management concerns surrounding high pressure membrane technologies, the EPA has adjusted RO/NF's technology projection compliance forecast to 0% in the EA. While the EPA does not estimate any water systems will elect to install RO/NF to comply with the PFAS rule, it remains a BAT for water systems to consider. For additional details on the EPA's EA, please see section XII.

The EPA also acknowledges that due to technical site-specific considerations, some BATs may not be the best choice for particular water types. PFAS treatment option selection should consider conditions for a given utility including water quality, available space, disposal options, and currently installed unit operations. AIX may be the preferred technology for some utilities based on expected treatment needs, while others may select GAC or other technologies. However, as many commenters indicated, the BAT designations are appropriate for water systems across the country.

Several commenters pointed out that GAC may release arsenic at levels exceeding arsenic's MCL temporarily when installed and upon changing media, deleteriously impacting finished water quality. While the EPA has documented challenges surrounding GAC and arsenic (USEPA, 2024l), the EPA disagrees that the arsenic release poses an exposure concern so long as appropriate procedures are followed. Those procedures include discarding the initial bed volumes (BVs) after installation or replacement of media. A bed volume is the volume of liquid contained within a GAC contactor, it is the container volume minus the solids volume and void space. The quantity of treated water discarded can be significant (*e.g.,* as high as 350 BVs as one commenter noted). However, this amount of discarded water is low in comparison to the normal service life between GAC replacement, which is approximately 84,000 BVs or approximately about 0.5% of the total treated volume. The total water volume discarded is also low in comparison to water loss through leaks across the United States, which account for about 15% of treated water or what would be approximately 12,600 BV equivalents for this system. While conserving water is a significant issue, the water discarded due to GAC applications is relatively low. Systems can reduce water discard associated with BAT implementation by using acid washed and/or prerinsed GAC or using buffered/pre-flushed resins for AIX. Any treatment technology can create problems if improperly maintained and operated. Finally, GAC has been statutorily designated as ''feasible for the control of synthetic organic chemicals,'' such as PFAS, in SDWA section 1412(b)(5).

The EPA received many suggestions for additional BATs including powdered activated carbon (PAC), alternative sorbents, and new destructive technologies. However, these alternative BATs proposed, except for PAC, currently lack demonstrated full-scale removal of the six PFAS under consideration. The EPA notes that there are some reports of PAC use on a temporary basis and that it can reduce PFAS concentrations in drinking water. PAC may be an appropriate choice of technology in certain circumstances, however, its efficacy for trace removal tends to be variable due to factors such as carbon particle size, background organics, and plant efficiency. Therefore, PAC is not as effective as GAC overall, and the agency has not designated it as a BAT. The EPA periodically reevaluates treatment technologies and may add additional technologies based on updated information. It is important to note that water systems may use any technology or practice to meet the PFAS MCLs and are not limited to the BATs in this rule. Other technologies may be chosen in lieu of BAT because they may be more cost effective or better suited to the specific operating conditions of the particular site to meet the MCL. Electing not to use a BAT, however, means that a system will not be eligible for a variance under SDWA section 1415(a)(1)(A). For example, if a facility does not install GAC where it is the designated BAT, but uses PAC instead, and fails to meet the MCL, the facility would not be eligible for a variance under SDWA section 1415(a)(1)(A). On the other hand, the same facility may be eligible for an exemption under SDWA section 1416 if, for example, GAC could not be installed due to an inability to obtain financing and PAC was used instead, and the facility failed to meet the MCL.

Many commenters pointed out the need for increased research, technological innovation, and guidance in treating drinking water for PFAS. The available information is sufficient to finalize the BATs as proposed but the EPA agrees that more research may be beneficial (USEPA, 2022c). With respect to the EPA's request for public comment on additional guidance materials that would be helpful to support successful technical implementation of the rule, the EPA received many comments related to the need for technical materials to support rule implementation. The agency plans to collaborate with states, technical assistance providers, industry associations and interested stakeholders, including small systems, following the rule promulgation to provide technical materials that can assist water systems in complying with the regulations. The EPA is currently funding many technical assistance efforts associated with PFAS, including supporting treatment infrastructure projects through the Drinking Water State Revolving Fund (DWSRF) and the Emerging Contaminant grant program as designated and funded through the Bipartisan Infrastructure Law (BIL).

Many commenters supplied information related to capital as well as operations and maintenance costs. Many commenters expressed concerns over potential costs and capacity while some commenters expressed the opposite opinion. These issues are further addressed in the EPA cost analysis in section XII and within the EPA's *Response to Public Comments on the Proposed PFAS NPDWR* (USEPA, 2024k). For additional discussion regarding the feasibility of the final MCLs, please see section V of this preamble above.

Many comments pointed to potential supply chain issues in both material and technical capacity such as qualified personnel, including certified operators. While there may be some supply chain issues in the short-term, comments from BAT suppliers indicate excess capacity as well as investment in production. Furthermore, while there may be temporary difficulties in supply chain and technical capacity, the structural demand increase is expected to lead to supply increases as well as innovation such as proposed technologies which were not designated as BATs. This has been historically demonstrated multiple times in prior drinking water rules. For example, activated alumina was listed as one of the BATs and a SSCT for arsenic removal in the Arsenic Rule (USEPA, 2001), and acknowledgement was given to granular ferric hydroxide media as a developing technology. While the granular ferric hydroxide media was not selected as a BAT/SSCT at the time due to lack of full-scale demonstration, these media became the predominant approach to addressing arsenic: Rubel (2003) stated that new iron-based materials could be ''employed economically on a spent media basis without the incorporation of pH adjustment chemicals and equipment.'' McCullough et al. (2005) cited over a dozen demonstration sites across the US implementing granular iron media treatment technologies, providing further supporting evidence that new technologies evolved in the wake of the Arsenic rule to provide more efficient and economical treatment systems. Additionally, the present statutory standard for ''best available technology'' under 1412(b)(4)(D) represents a change from the provision prior to 1986, which required the EPA to judge feasibility on the basis of ''best technologies generally available'' (BTGA). The 1986 Amendments to the SDWA changed BTGA to BAT and added the requirement that BAT must be tested for efficacy under field conditions, not just under laboratory conditions. The legislative history explains that Congress removed the term ''generally'' to assure that MCLs ''reflect the full extent of current technology capability'' [S. Rep. No. 56, 99th Cong., 1st Sess. at 6 (1985)]. Read together with the legislative history, the EPA has concluded that the statutory term ''best available technology'' is a broader standard than ''best technology generally available,'' and that this standard allows the EPA to select a technology that is not necessarily in widespread use, as long as its performance has been validated in a reliable manner. Indeed, the 1991 Lead and Copper Rule stated, ''as long as it has been tested beyond the laboratory under full-scale conditions for other contaminants, and the performance of the technology for lead and copper may reasonably be projected based upon other available treatment data (*i.e.,* laboratory or pilot scale), the EPA believes the technology can be established as BAT.''

With respect to the challenges raised by commenters surrounding capital improvement, the EPA has provided compliance flexibility by providing a two-year capital improvements extension of the MCL compliance deadline allowed by section 1412(b)(10) of SDWA. Additionally, the EPA will continue its research as well as outreach efforts to help develop technical and operator capacities. For comments and additional information regarding the implementation timeframe for this rule, please see section XI.D.

Many commenters stated that permitting needs to be streamlined and that more assistance should be proffered to primacy agencies, utilities, and other interested stakeholders. While SDWA does not require permits, state and local authorities often require permits for the installation of treatment facilities at water systems. The EPA has developed supporting rule documents such as the *Best Available Technologies and Small System Compliance for Per- and Polyfluoroalkyl Substances (PFAS) in Drinking Water* document (USEPA, 2024l) that can be used to help permitting authorities develop more familiarity with these technologies over time. After finalization of the PFAS National Primary Drinking Water Rule (NPDWR), the EPA also intends to work with stakeholders to provide support to utilities, primacy agencies, and other interested parties to ensure successful rule implementation.

3. Final Rule

In the final rule, the EPA is codifying GAC, AIX, NF, and RO as BATs. The record does not support including additional BATs at this time. A BAT designation is informational, and while installation of the BAT is a condition of a variance under section 1415(a)(1)(A), systems without a variance are not required to use a BAT for MCL compliance. The owner/operator of a PWS will need to consider site specific circumstances as well as technical, economic, and local regulatory considerations when choosing a compliance technology for this rule. To address the challenges raised by commenters surrounding capital improvement, the EPA has provided a two-year compliance extension for capital improvements which is discussed in greater detail in section XI (*Rule Implementation and Enforcement*) and will continue its research efforts. The two-year capital improvement extension should also provide time for development of technical capacities and qualified personnel including certified operators. In response to public comment and in acknowledgement of residuals management concerns surrounding high pressure membrane separation technologies, the EPA is lowering RO/NF's technology projection compliance forecast in the EA. For comments and additional information related to the EPA's cost analysis, please see section XII. For comments and additional information regarding the implementation timeframe for this rule, please see section XI.D.

B. PFAS Co-Removal

1. Proposal

The EPA stated that AIX and GAC are effective at removing PFAS and there is generally a linear relationship between PFAS chain length and removal efficiency shifted by functional group. The EPA also notes that perfluoroalkyl sulfonates (PFSA), such as PFOS, are removed with greater efficiency than the corresponding perfluoroalkyl carboxylic acid (PFCA), such as PFOA, of the same carbon backbone length. Additionally, the compounds with longer carbon chains display a smaller percentage decrease in average removal efficiency over time (McCleaf et al., 2017). These same technologies also remove other long-chain and higher carbon/higher molecular weight PFAS as well as total organic carbon (TOC, DBP precursors). RO and NF may also remove other contaminants including arsenic, TOC, and chromium-VI. In short, the EPA noted that this regulation, if finalized, would result in a reduction of the six PFAS proposed for regulation, other co-occurring PFAS, and other co-occurring contaminants.

2. Summary of Major Public Comments and EPA Responses

A significant majority of commenters supported the EPA's position that treatment technologies which remove PFAS provide ancillary benefits by removing other known or potential contaminants. One commenter disputed the ability of these technologies to provide ancillary benefits, and others suggested that the EPA's proposed regulation would provide only limited protection against the many PFAS not under consideration in the rule. The EPA disagrees with the commenters who state that the proposed regulation would not result in a reduction in co-occurring PFAS and other contaminants. Burkhardt et al. (2023) used a theoretical approach[11] to estimate that all but one of the PFAS that are quantified by EPA Methods 533 and 537.1 could be economically removed by GAC in typical water qualities and that of 428 PFAS evaluated, 76–87 percent could be cost-effectively treatable. The co-removal benefits are well documented in the scientific literature and in the evidence submitted by public comment. The *Best Available Technologies and Technologies and Cost* support documents summarize literature demonstrating the co-removal capabilities of treatment technologies.

Some commenters stated that treatment for one PFAS does not inherently imply removal of other PFAS. The EPA agrees, as discussed in the proposed rule preamble. In general, there is an inverse relationship between treatability and toxicity which is tied to the carbon backbone (Bellia et al., 2023). Generally, the longer the carbon backbone length, the more easily the PFAS is removed by a given treatment technology. For example, if PFOA (C8) is targeted for removal by the water system, perfluorodecanoic acid (PFDA, C10) would most likely be removed as well. However, the converse would not

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[11] While PFAS are often discussed as a group, the individual PFAS species can have a range of different removal efficacies using GAC. A theoretical approach for PFAS fills information gaps where analytical methods do not exist for all PFAS and testing is expensive and time consuming

be true (*i.e.,* a system targeting PFNA (C9) removal would reduce PFHxA (C6) to a lesser extent).

Some commenters suggested that co-removal would decrease the removal efficiency of GAC or AIX and that removal efficiency of non-target contaminants is lower than it could otherwise be. The EPA agrees that the removal of non-targeted contaminants by GAC or AIX can lower the PFAS removal efficiency; the agency has accounted for this uncertainty in appendix N of the EA (USEPA, 2024e). The EPA also agrees that targeting contaminants for removal will be more effective than relying on other non-targeted removal. For example, a GAC facility designed to remove PFAS will not be as effective at removing DBP precursors as a facility designed for that; however, there will still be co-removal of DBP precursors which may lead to a reduction in DBPs. Ultimately, treatment facilities operate best when tailored to specific contaminants or mixture of contaminants unique to that location. For additional information on the EPA's co-benefit analysis, please see section XII.

Some commenters expressed concern about co-removal taking beneficial ions from water, specifically fluoride ions, and suggested that would be an added cost to the rule. The EPA notes that fluoride has a legally enforceable MCL of 4.0 mg/L, and a non-enforceable secondary standard of 2.0 mg/L to prevent mild or moderate dental fluorosis. The EPA also notes that while some PFAS do contain organic fluorine bound to carbon, fluorine and fluoride are not the same. The BATs identified for the removal of PFAS for drinking water are not optimized for the removal of fluoride and do not necessarily provide effective removal of naturally occurring fluoride. For example, GAC is ineffective for fluoride removal at environmentally relevant pHs (USEPA, 2024o).

Some commenters suggested that co-removal may make it more difficult to dispose of materials left over from the drinking water treatment processes, known as treatment residuals. For example, GAC may remove and concentrate radon or other contaminants to such an extent that the spent media is considered hazardous. The EPA believes that removing hazardous constituents from drinking water is generally beneficial even though it could complicate residual management. More details on treatment residuals, are discussed in part C of this section.

Some commenters also suggest more research may be beneficial to understanding co-removal. The EPA agrees (USEPA, 2022c).

3. Final Rule

GAC, AIX, NF, and RO are codified in the final rule as BATs. As discussed elsewhere in the record for this final rule, because of PFAS co-occurrence and the ability for treatment technologies to co-remove co-occurring PFAS and other contaminants, the EPA anticipates the final rule will result in significant co-removal public health benefits in addition to those benefits from removing the six PFAS being directly regulated by this action.

C. Management of Treatment Residuals

1. Proposal

As part of the BAT evaluation, the EPA reviews full-scale studies that fully characterize residual waste streams and disposal options. The EPA found that the most likely management options for spent material containing PFAS is reactivation for GAC, incineration for spent IX resin, and for disposal of RO/NF retentate, treatment and discharge via a NPDES compliant facility to surface water or, sanitary sewer, or in limited circumstances, underground injection. Large volumes of spent GAC and AIX containing PFAS are periodically generated and must be removed which does not lend itself to on-site storage over time. The EPA stated that the disposal options identified in the 2020 *Interim PFAS Destruction and Disposal Guidance* (USEPA, 2020d) are landfill disposal, thermal treatment, and in limited circumstances, underground injection.

The EPA recognizes that future actions through statutory authorities other than SDWA may have direct or indirect implications for the residuals from drinking water treatment. Future hazardous waste listings for certain PFAS may limit disposal options for spent drinking water treatment residuals containing PFAS and/or potentially increase costs. A CERCLA designation as a hazardous substance does not restrict, change, or recommend any specific activity or type of waste (USEPA, 2022l). The EPA evaluated the potential impact on PWS treatment costs to PWSs associated with hazardous residual management should PFAS be listed as a hazardous waste in the future. For comments and additional information related to the EPA's cost analysis, please see section XII.

2. Summary of Major Public Comments and EPA Responses

While some commenters stated that more research can be beneficial to further our understanding of managing PFAS treatment residuals, others urged the EPA to proceed with this rulemaking as expeditiously as possible in the interest of public health. Others argued that the EPA should delay this action until the *PFAS Destruction and Disposal Guidance* is updated. The National Defense Authorization Act for Fiscal Year 2020, Public Law 116–92, section 7361, directs the EPA to revise the *PFAS Destruction and Disposal Guidance* triennially; the new destruction and disposal guidance is anticipated to be released approximately concurrently with this rule and further revisions may be expected before the effective dates for this rule. The EPA disagrees that the projected significant and direct public health protections for drinking water consumers in this rule should be delayed for the revision of guidance on management of PFAS waste streams.

Many commenters expressed concern that not enough was being done to manage spent drinking water treatment residuals containing PFAS at the end of their useful working life and that residual management amounted to media shifting (*i.e.,* taking PFAS from water via sorption media then landfilling that media does nothing to reduce the overall amount of PFAS). Many commenters stated that landfills and thermal treatment facilities can potentially be PFAS sources as the BATs in this rule are separative as opposed to destructive technologies.

The EPA notes that from a mass balance perspective, PFAS removal from drinking water is generally anticipated to result in lower concentrations of PFAS in the environment. With appropriate controls, landfills, and thermal treatment of PFAS contaminated media can minimize PFAS releases to the environment (USEPA, 2020d). Sorptive media can be incinerated or reactivated. There is also ongoing research into destructive and sequestration technologies that may help quantify the extent to which PFAS may be destroyed some of which is funded by the EPA (USEPA, 2022c).

Furthermore, it is also important to distinguish between a potential environmental release and a direct exposure. A PFAS release does not inherently imply human exposure and a release is not inherently risky to specific populations. From a risk management perspective, while the EPA acknowledges that while each destruction and disposal technology has limitations, a potential environmental release under point source management is anticipated to be a more health protective alternative than human exposure through drinking water.

Some commenters recommended the EPA consider additional destruction and disposal technologies. The EPA notes that disposal and destruction technologies are currently available to manage drinking water residuals. The EPA appreciates the example destructive technologies, and while beyond the scope of finalizing this NPDWR, the agency intends to consider additional destruction and disposal technologies in future destruction and disposal guidance.

Many commenters, including destruction and disposal trade associations, stated there would be difficulties managing spent residuals containing PFAS generated from drinking water treatment. In contrast, other commenters stated that there was existing national capacity and at least one company stated they were actively evaluating investment for additional capacity to handle residuals. The record demonstrates that there is existing national capacity to handle spent drinking water residuals containing PFAS in a manner that minimizes risk to human health. Destruction and disposal of PFAS-containing materials is currently not subject to certain hazardous waste regulation and therefore the materials may be managed in non-hazardous and hazardous waste treatment and disposal systems (USEPA, 2020d). Hazardous waste is regulated pursuant to RCRA authority 42 U.S.C. 6921–6939 (also known as RCRA ''Subtitle C''). The regulatory definition of hazardous waste is found in 40 CFR 261.3. PFAS are currently not a listed hazardous waste or characterized as a hazardous waste, but a PFAS-containing waste may meet the regulatory definition of hazardous waste if PFAS is mixed with a listed hazardous waste or if a PFAS-containing mixture exhibits a hazardous characteristic (*e.g.,* corrosivity or another characteristic stemming from the material that is mixed with PFAS). PFAS which are commingled with hazardous substances and/or hazardous wastes will be subject to the appropriate rules and regulations and may be included as Applicable or Relevant and Appropriate Requirements on a site-specific basis. Not all disposal sites may be appropriate for spent drinking water treatment residuals containing PFAS and the EPA strongly encourages owners and operators of treatment facilities to refer to appropriate and up-to-date guidance on treatment residual management such as the *2020 Interim Guidance on the Destruction and Disposal of Perfluoroalkyl and Polyfluoroalkyl Substances and Materials Containing Perfluoroalkyl and Polyfluoroalkyl Substances* (USEPA, 2020d) and subsequent updates.

The EPA anticipates approximately 226,500 short tons of spent drinking water media such as activated carbon and AIX resin to be generated annually as a result of this rule; in calendar year 2018 alone, the U.S. generated about 290 million short tons of waste (USEPA, 2022m). The increase in total waste caused by this action is approximately 0.08% of the total U.S. waste produced. This is a minor change in aggregate waste produced; the same amount as a pound contributes to a ton. Even if PFAS were to be designated in the future as regulatory hazardous waste, there is existing capacity to handle these waste streams through existing hazardous waste facilities in every state. Some water systems may have to ship hazardous wastes significant distances; however, the main cost driver is disposal fees not transportation. The EPA rejects the assertion that it has not evaluated if sufficient capacity exists for disposal and storage of PFOA and PFOS contaminated materials. The EPA also acknowledges that CERCLA section 104(c)(9) does not allow the agency to initiate a remedial action, unless the state first enters into a state Superfund State Contract or Cooperative Agreement (CA) that assures the availability of adequate capacity to manage hazardous wastes generated in the state for 20 years following the date of the response agreement. The EPA's rulemaking designating PFOA and PFOS as CERCLA hazardous substances, if finalized, does not impose any capacity concerns that require further action under section 104(c)(9). In that action, the EPA is designating PFOA and PFOS as CERCLA hazardous substances. No PFAS are currently listed, or being proposed to be listed, as hazardous wastes under RCRA. The 2021 *Biennial Report Summary Results* indicate about 18 million tons of hazardous wastes are normally generated annually. Drinking water treatment materials then would constitute about a 1.26% increase in hazardous wastes generated annually. Since there is over twenty years' capacity, the relatively small magnitude of the increase indicates that waste management capacity is sufficient in the short term should PFAS be designated as regulatory hazardous wastes.

Many commenters conveyed concern over the cost of drinking water residuals management resulting from finalizing this rule. The EPA conducted an EA to help address these concerns. For comments and additional information related to the EPA's cost analysis, please see section XII.

While no PFAS are currently listed as regulatory hazardous wastes under RCRA, in response to stakeholder feedback, the EPA included a sensitivity analysis to determine the impact on water systems should they be required to handle and dispose of PFAS treatment materials as hazardous waste in the future. The results of this analysis can be found in the EA for this rule (USEPA, 2024g). Some commenters suggested that accounting for future potential regulations is uncommon, and trying to account for all potential future contingencies would make economic analyses impossible. The EPA strongly agrees and has not attempted to do so here; this analysis was limited to looking at a hypothetical future hazardous waste listing situation because that has been of particular concern in this rule. Some commenters stated that the EPA should account for the public health benefits of treating PFAS as hazardous wastes, not just additional costs incurred. The EPA agrees and has modified the analysis to include a qualitative statement about the public health benefits which could potentially arise from treating PFAS as hazardous wastes. Many commenters stated that the EPA hazardous waste cost would drive the total cost higher than the 3–5% estimated by the EPA. After considering public comment, the EPA has revised the final cost estimates in this rule. The EPA estimated increased cost would be approximately $99M at the 2% discount rate. The increased cost was driven by updating the dollar year of cost curves from 2021 to 2022 which increased waste management unit costs by approximately 12%; implementing a cap on media life even if not indicated; changing the technology compliance forecast by eliminating RO/NF while increasing GAC and AIX (thereby increasing spent media volume); and increasing occurrence estimates for the final rule compared to the proposed rule, triggering more systems into treatment. The increased costs were not driven by changes to unit cost estimates for hazardous waste management. The EPA believes its assessment is accurate; the total cost encompasses capital costs, maintenance, design, and operations, including waste management. Waste management costs are thus a subset of operational cost which in turn is a subset of total costs; generally, changes in the cost of one subcomponent would not significantly influence total costs, and the record does not reflect that a change in waste disposal costs would have a significant impact on total costs under this rule. These estimates are discussed in greater detail in the HRRCA section of this rule and in appendix N of the EA (USEPA, 2024e).

Many commenters suggested that regulations under other statutes, particularly a potential CERCLA hazardous substance designation, will increase disposal costs. The EPA disagrees that, if finalized, the CERCLA hazardous substance designation for PFOA and PFOS will increase disposal costs for water treatment facilities. The designation of PFOA and PFOS as CERCLA hazardous substances would not require waste (*e.g.,* biosolids, treatment residuals, etc.) to be treated in any particular fashion, nor disposed of at any specific particular type of landfill. The designation also does not restrict, change, or recommend any specific activity or type of waste at landfills. Along with other release notification requirements, CERCLA designation would require that any person in charge of a vessel or facility report a release of PFOA and/or PFOS of one pound or more within a 24-hour period. The EPA does not expect spent drinking water treatment residuals containing PFAS to be released into the environment at or above the reportable quantity as a part of standard residuals management practices used by water systems. This is because the PFAS loading onto sorptive media is very small. The weight percent of PFAS onto GAC under normal treating scenarios will vary widely; however, a reasonable order of magnitude estimate is 1×10^{-5} grams PFAS per gram of sorbent in full-scale applications. High pressure membranes split water into a treated stream and concentrated waste stream. The concentrated waste stream will contain about 5–12 times more PFAS than the influent which is likely to still be in the ng/L scale. A drinking water facility which takes reasonable precautions is unlikely to release enough low concentration residuals to release one pound of PFOA and/or PFOS within a 24-hour period. At the concentrations discussed above, to exceed a one-pound threshold, a facility using sorptive techniques would have to release approximately 50 tons of sorbent, within a 24-hour period. A one-pound uncontrolled release from RO or NF facilities, assuming 500 ng/L of PFAS in the reject water, would require approximately 240 million gallons of high-pressure membrane concentrate to be released within 24 hours. Additionally, neither a release nor a report of a release automatically requires any response action under CERCLA. The EPA makes CERCLA response decisions based on site-specific information, which includes evaluating the nature, extent, and risk to human health and/or the environment from the release. Hazardous substance designations do not automatically result in CERCLA liability for any specific release. Whether an entity may be subject to litigation or held liable under CERCLA are site-specific and fact-dependent inquiries. Likewise, CERCLA affords the Federal Government broad discretion as to whether or how to respond to a release. For those reasons, the EPA cannot assess with reasonable certainty what litigation or liability outcomes may indirectly result from this designation since those outcomes are often linked to the EPA's discretionary decisions with respect to CERCLA response actions as well as site-specific and fact-dependent court rulings.

Many commenters suggested that high pressure membranes, which separate PFAS from one stream and concentrate it in another stream, may not be feasible as a BAT because utilities treating and discharging reject water from high pressure membranes typically require a NPDES permit. The EPA disagrees because there are currently full-scale facilities which use this technology to treat PFAS and high-pressure membranes may be the best viable option in a multi-contaminant setting. The brine may undergo further pre-treatment as part of a process train to enable discharge, such as GAC or AIX treatment. Some RO/NF applications discharge directly to surface water or through an interconnection to a wastewater treatment plant. The EPA, however, does agree that brine treatment or disposal may be challenging and in 2022, the EPA issued memorandum that recommended NPDES and POTW pretreatment program permitting conditions for PFAS discharges (USEPA, 2022d; USEPA, 2022e). In conclusion, in limited applications, high pressure membranes may still serve as a viable treatment strategy, such as for facilities with access to brine treatment or disposal.

Some commenters suggested that reactivation was not permissible under the *2020 Interim PFAS Destruction and Disposal Guidance* or that interim storage was required. Commenters are incorrect in their interpretation of the plain language in that guidance. The guidance does not state that reactivation or thermal treatment are prohibited. The guidance does acknowledge a need for further refinement and research and that interim storage may be an option if the immediate dispensation of PFAS-containing materials is not imperative. However, nowhere does that guidance mandate interim storage or prohibit other forms of PFAS destruction and disposal.

3. Final Rule

The final rule does not specifically require any specific destruction or disposal practices for spent media containing PFAS. The EPA has considered residual waste streams and disposal options and found that management options exist for treatment residuals containing PFAS.

D. What are Small System Compliance Technologies (SSCTs)?

1. Proposal

Section 1412(b)(4)(E)(ii) requires that the agency identify SSCTs, which are affordable treatment technologies, or other means that can achieve compliance with the MCL. The EPA identified SSCTs using the affordability criteria methodology developed for drinking water rules (USEPA, 1998b) and proposed the following table which shows which of the BATs listed above are also affordable for each small system size category listed in section 1412(b)(4)(E)(ii) of SDWA.

Table 23: Proposed SSCTs for PFAS Removal

System Size (Population Served)	GAC	IX	RO/NF	Point of Use (POU) RO/NF [1]
25-500	Yes	Yes	No	Yes
501-3,300	Yes	Yes	No	Yes
3,301-10,000	Yes	Yes	Yes	not applicable [2]

Notes:

[1] POU RO is not currently listed as a compliance option.

[2] Implementing and maintaining a large-scale POU program is likely to be impractical.

Point-of-use (POU) and point-of-entry (POE) were not listed as compliance options because the regulatory options under consideration require treatment to concentrations below the current NSF International/American National Standards Institute (NSF/ANSI) certification standard for POU device removal of PFAS. As the EPA has determined that affordable SSCTs are available, the agency is not proposing any variance technologies.

2. Summary of Major Public Comments and EPA Responses

Many commenters stated that the POU/POE water treatment industry may already have multiple products that can reduce PFAS chemicals to below the proposed MCL. Additionally, some commenters stated that the influent used (*i.e.,* the challenge water) to test these POU/POE products often contains much higher concentrations of PFAS than would normally be found in most source waters. Commenters also pointed out that under NSF/ANSI, 53 and 58 certifications exist for total PFAS (PFOA, PFOS, PFHxS, PFHxA, and PFDA), as well as PFHpA, PFHxS, and PFNA individually. However, SDWA section 1412(b)(4)(E)(ii) requires that SSCTs achieve compliance with the MCL or treatment technique. While devices certified to the NSF/ANSI standards must be demonstrated to significantly reduce PFAS concentrations and, in many cases, can reasonably be expected to treat below this rule's MCLs, the current standards and certification procedures do not assure compliance with this rule. In particular, PFBS and HFPO–DA, have no certification standards at this time and the certification standards for PFOA, PFOS, and PFHxS are above this rule's MCL. The certification standards for PFOA, PFOS, and PFHxS are 20 ng/L, compared to the MCLs of 4.0 ng/L for PFOA and PFOS, as well as 10 ng/L for PFHxS; the total PFAS certification standard is 20 ng/L effluent comprised of PFOA, PFOS, PFHxS, PFHxA, and PFDA compared to a Hazard Index of 1 for mixtures of PFHxS, PFNA, HFPO–DA and PFBS. Since the NPDWR has standards that NSF/ANSI are currently unable to verify, POE/POU technologies could potentially not achieve compliance contrary to SDWA section 1412(b)(4)(E)(ii) which requires that SSCTs achieve compliance with the MCL. While POU/POE technologies may provide significant levels of protection, and the EPA anticipates they will eventually comply with the NPDWR, there is not yet a systematic verification process in place for the level of protection provided by these devices. As mentioned in the proposal, the EPA is aware that the NSF/ANSI Drinking Water Treatment Unit Joint Committee Task Group is in the process of updating their standards; should these future standards meet the NPDWR, the EPA could revise the SSCT list to include POE/POU.

Many commenters also correctly pointed out numerous challenges surrounding POU/POE as a compliance option for some PWSs such as resident cooperation, operation and maintenance, monitoring, and implementation of distributed treatment approaches. The EPA agrees implementation of POU/POE as a compliance option for any NPDWR can be challenging for some PWSs but also agrees with commenters who noted that POU/POE can provide flexibility and compliance options to very small water systems or certain NTNCWS such as schools, factories, office buildings, and hospitals that provide their own water.

The EPA received many comments that other POU devices other than RO/NF should be acceptable ways to meet the MCLs for small systems. For instance, commenters noted that a combination GAC/AIX device with filters could reduce PFAS concentrations to below the MCL values. The EPA agrees and has changed wording in the final rule preamble and related supporting documents that implied that only RO/NF POU devices would be able to meet a future certification standard. The EPA notes that for small systems, as long as the proposed POU/POE devices are certified by an appropriate third-party certifier (*e.g.,* ANSI/NSF) to meet the regulatory MCL, they would meet the requirements of this regulation. The EPA also received many requests to change the way data was displayed in tables 20 and 22 of the proposed rule which summarized proposed SSCTs for PFAS removal and total annual cost per household for candidate technologies. In the proposal, the EPA wrote that this data was "Not Applicable" because of the economies of scale for centralized treatment. While the EPA still believes that a POU program that large is likely to be impractical, the EPA has changed the way this is displayed by replacing the term "Not Applicable" with "Data Unavailable." The EPA notes that neither of these changes imposes nor relieves any rule requirements and only serve to recharacterize the way the EPA reports available technologies.

The EPA asked for comment on the national level analysis of affordability of SSCTs and specifically on the potential methodologies presented in the EA for the proposed rule section 9.12. A couple of commenters recommended the EPA not use median household income (MHI) in the affordability analysis. The EPA decided to retain the MHI measure of income in its primary national level SSCT affordability methodology, and specifically use 2.5% of the MHI as the affordability threshold, given the value is easily understandable and available, providing a central tendency for income which is representative of a whole community's ability to pay and is not unduly influenced by outlier values. However, in this rule, the EPA recognizes the value in examining alternative measures of a community's ability to afford an SSCT, so the agency chose to include supplemental analyses that use alternative metrics, specifically 1% of MHI, 2.5% of lowest quintile income (LQI), and an analysis accounting for financial assistance. See chapter 9.13.2 of the EA for more details. These supplemental analyses help to characterize affordability when considering the marginal impact, disadvantaged community groups, and subsidization.

Some commenters stated that the data the EPA used to inform current water rates from the 2006 Community Water System Survey (CWSS) is outdated. While dated, the data from the 2006 CWSS remains the best available dataset for this national level analysis and affordability determination for the following reasons: (1) the CWSS survey used a stratified random sample design to ensure the sample was representative and (2) these responses can be extrapolated to national estimates since the survey has a known sampling framework; and the data can be organized by system size, source, and ownership (USEPA, 2020e).

Some commenters recommended the EPA extend the affordability analysis to medium and large systems. The EPA disagrees with this recommendation, as the purpose of this analysis is to determine if available SSCTs are affordable, per SDWA section 1412(b)(4)C(ii). Therefore, the EPA chose to continue to analyze small system technologies rather than include medium and large systems.

Some commenters specifically disagreed with one of the EPA's supplemental affordability analyses that examined the impact of the rule when accounting for the financial assistance through BIL and other sources that are generally available to small systems. These commenters stated that the EPA should not assume that this funding will be available or enough to cover the small system capital costs associated with the rule. The EPA conducted this supplemental analysis in response to the recommendations of the SAB, which stated, ''[i]f this funding is readily available to many or most systems facing affordability problems, it seems appropriate to take the availability of this funding into account in determining national level affordability.'' (USEPA, 2002b) The EPA disagrees with these commenters as this significant funding will be generally available, and the EPA continues its efforts to help PWSs access it. It is therefore reasonable to consider the burden reduction in the supplemental affordability analysis.

Some commenters disagreed with the EPA's affordability determination because they stated it was based on inaccurate treatment cost information. A couple of commenters presented their own estimates for small system household costs and compared these estimates to the EPA's affordability threshold and concluded the rule is unaffordable. The EPA disagrees with many of the underlying assumptions in the commenters' cost estimates which, on whole, result in overestimated household costs, see section XII.A. These commenters cited cost information that is not representative of the range of treatment costs nationally, and the EPA disagrees with the commenter's cost model that systematically overestimates capital operation and treatment costs. The EPA updated the affordability analysis for the national affordability determination using the updated treatment cost curves (discussed in section XII.D) and found for systems serving between 25 and 500 people, that the upper bound estimated annual household treatment costs for GAC exceed the expenditure margin. Lower bound estimated annual household treatment costs for GAC do not exceed the expenditure margin; for more information see section XII. These exceedances are primarily driven by capital costs and attributable to the use of high-cost materials (*e.g.,* stainless steel) in the upper bound estimates. Systems using low-cost materials, but with source water characteristics otherwise set to the upper bound (*e.g.,* influent PFAS at approximately 7,000 ng/L, influent TOC at 2 mg/L), would fall below the expenditure margin. Although costs increase in some scenarios, the increases are not significant enough to change the conclusions about affordability. The small system compliance technologies available to meet the requirements of the final rule are affordable for all small systems when the technologies do not use the high-end materials. Technologies that do not use high end materials may be less durable but nonetheless are available for small systems and can meet the requirements of the final rule. For more information on the EPA's response to comments on treatment costs see section XII. The EPA also disagrees that there are no affordable compliance technologies for small systems as the EPA has demonstrated that SCCTs are available below the affordability threshold using the best available peer reviewed information to support the agency's cost estimates.

3. Final Rule

The final rule includes sorptive devices as well as combination devices, should they meet third party verification standards and the MCL. In USEPA, 2024l, the EPA also changed the way data are presented by replacing the term ''Not Applicable'' with ''Data Unavailable'' in response to public comment. Finally, the final affordability analysis reflects updates made to the unit cost curves after considering public comments. The EPA has determined that affordable SSCTs are available that meet the requirements of the final rule (see table 6 to paragraph (e) of 40 CFR 141.61).

The EPA's affordability determination for the final rule, using long standing EPA methodology and supplemental affordability analyses can be found in the EA chapter 9.12.

The EPA notes that POU RO devices are not currently listed as a SSCT because the NPDWR requires treatment to concentrations below the current NSF International/American National Standards Institute (NSF/ANSI) certification standard for POU device removal of PFAS. However, POU treatments are reasonably anticipated to become a compliance option for small systems in the future if NSF/ANSI develop a new certification standard that mirrors or is more stringent than the final regulatory standards. Other third-party entities including NSF can independently certify drinking water treatment units (DWTUs) that meet these standards. NSF/ANSI is considering lowering its current standard to levels closer to final standards in this NPDWR. Based on efficacy of reverse osmosis technology, RO POU devices can reasonably be anticipated to remove the majority of PFAS when they are properly designed and maintained. Other POU devices (*e.g.,* activated carbon) may also meet future EPA PFAS regulatory limits. These devices would also need third-party testing and certified against the regulatory standards. Further, the EPA notes that water systems may use any technology or practice to meet the MCLs promulgated in this NPDWR and are not limited to the BATs nor SSCTs discussed in this section. Other technologies or nontreatment options may be chosen in lieu of a BAT or SSCT because they may be more cost effective or better suited to the specific operating conditions of the particular site to meet any MCL.

XI. Rule Implementation and Enforcement

A. What are the requirements for primacy?

1. Proposal

SDWA section 1413 establishes requirements that primacy agencies (states, Tribes and territories) must meet to have primary enforcement responsibility (primacy) for its PWSs. These include: (1) adopting drinking water regulations that are no less stringent than Federal NPDWRs in effect under sections 1412(a) and 1412(b) of SDWA; (2) adopting and implementing adequate procedures for enforcement; (3) keeping records and making reports available on activities that the EPA requires by regulation; (4) issuing variances and exemptions (if allowed by the state) under conditions no less stringent than allowed by SDWA sections 1415 and 1416; and (5) adopting and being capable of implementing an adequate plan for the provision of safe drinking water under emergency situations. The regulations in 40 CFR part 142 set out the specific program implementation requirements for states to obtain primacy for the Public Water System Supervision (PWSS) Program, as authorized under section 1413 of the Act.

Under 40 CFR 142.12(b), all primacy agencies are required to submit a revised program to the EPA for approval within two years of promulgation of any final PFAS NPDWR or request an extension of up to two years in certain circumstances. To be approved for a program revision, primacy agencies are required to adopt revisions at least as stringent as the revised PFAS-related provisions. To obtain primacy for this rule, primacy applications must address the general requirements specified in subpart B of part 142. The EPA proposed special primacy requirements for the PFAS NPDWR (§ 142.16(r)), to outline additional requirements for a primacy agency related to identifying its plan for implementing the initial monitoring requirements.

2. Summary of Major Public Comments and EPA Responses

The EPA received one comment that most of the initial monitoring may occur before primacy applications will be submitted, which are not due until two years after final rule promulgation. A couple of commenters assert that it is unclear why states are required to include an initial monitoring plan in their primacy application and that states will not be able to implement and demonstrate that this monitoring plan is enforceable under state law until state regulations have been promulgated. The EPA recognizes that some initial monitoring by water systems may occur prior to a state, territory, or Tribe receiving the EPA approval for primacy and agrees with the commentor that for states to develop a monitoring plan that addresses when systems will be scheduled to conduct initial monitoring is not a necessary requirement for a primacy application. However, where states are approved for primacy before the compliance date for the water systems, primacy agencies should have procedures for evaluating whether data that a CWS or NTNCWS submits to satisfy the initial monitoring requirements are acceptable. It is therefore appropriate to require primacy agencies to include in their primacy application a description of their procedures for reviewing water system's use of pre-existing data to meet initial monitoring requirements, including the criteria that will be used to determine if the data are acceptable and the primacy agency's procedures for ensuring water system compliance within the required timeframes. The compliance deadline for this initial monitoring by systems is three-years from promulgation, by which time primacy agencies should have primacy or interim primacy. To address the possibility that a state, Tribe, or territory may get an extension to apply for primacy, the final rule provides that these special primacy requirements are not applicable after the initial monitoring deadline (*i.e.,* three years after publication of the rule in the **Federal Register**). When a primacy agency does not yet have primacy for a new drinking water rule, an NPDWR is nonetheless applicable to water systems and may be enforced by the EPA following the compliance dates specified in § 141.900(b).

3. Final Rule

The EPA is revising the requirements for primacy as proposed in 40 CFR 142.16(r) by removing the requirements to develop an initial monitoring plan, although the EPA is finalizing the proposed requirement for primacy agency procedures for ensuring all systems complete the initial monitoring period requirements, including for determining whether pre-existing data are acceptable, but clarifying that these requirements would not apply after the deadline for initial monitoring has passed (*i.e.,* three years after publication of the rule in the **Federal Register**). The EPA also corrected two grammatical errors. In the final rule, the EPA requires that a PWS complete the initial monitoring by three years following date of promulgation (for additional discussion on monitoring and compliance requirements, please see section VIII of this preamble). It is the EPA's expectation that primacy agencies will have completed the requirements for primacy within the two years (*i.e.,* without an extension) and in that case, they will have the authority in place to ensure that systems comply with the initial monitoring requirements. If a primacy agency is applying for primacy after the deadline for initial monitoring has passed, then the requirement is no longer applicable. In that case, an NPDWR is nonetheless applicable to water systems and implementation would be overseen and enforced by the EPA consistent with any agreements with the state pursuant to the primacy application extension approval.

B. What are the record keeping requirements?

1. Proposal

The current regulations in 40 CFR 142.14 require primacy agencies to keep records of analytical results to determine compliance, system inventories, sanitary surveys, state approvals, vulnerability and waiver determinations, monitoring requirements, monitoring frequency decisions, enforcement actions, and the issuance of variances and exemptions. The primacy agency record keeping requirements remain unchanged and would apply to PFAS as with any other regulated contaminant.

2. Summary of Major Public Comments and EPA Responses

The EPA received a few comments about the record keeping that primacy agencies must maintain for compliance determinations and reporting, storing PWS facility data, tracking monitoring schedules, and keeping the public informed of the quality of their drinking water. As noted in the comments, most primacy agencies rely on SDWIS, developed by the EPA, to support this record keeping requirement. It was recommended that the EPA develop a data system, either SDWIS or a replacement, that is capable of fully managing the data associated with the proposed rule. Further, it was recommended that the EPA develop data management solutions such as a mechanism for migrating UCMR data into SDWIS State to reduce or eliminate the burden of ensuring compliance with the initial monitoring. The EPA agrees that appropriate data management solutions are needed to effectively comply with SDWA requirements; however, the agency does not believe these systems must be available at the time of rule promulgation. Additionally, while beyond the scope of this rulemaking itself, the EPA is actively working on PFAS data management solutions, including DW–SFTIES support and potentially updating the SDWIS suite of applications to manage data reported from this rule.

3. Final Rule

The primacy agency record keeping requirements in 40 CFR 142.14 remain unchanged and would apply to PFAS as with any other regulated contaminants. Water system recordkeeping requirements are referenced within subpart Z in § 141.904. In the final rule, the EPA updated this regulatory text to cross-reference the record retention provisions in § 141.33. The EPA is developing the Drinking Water State-Federal-Tribal Information Exchange System (DW–SFTIES) that will support all SDWA drinking water rules. The EPA plans to continue to provide support for necessary updates to SDWIS State, including for reporting requirements for new rules, until the DW–SFTIES is in production and in use by primacy agencies. SDWIS State support and updates will continue until the DW–SFTIES Board recommends a sunset date after DW–SFTIES is in production and in use by primacy agencies. The EPA will evaluate the migration of UCMR data into the suite of SDWIS applications.

C. What are the reporting requirements?

1. Proposal

Under 40 CFR 142.15, primacy agencies must report to the EPA information regarding violations, variances and exemptions, enforcement actions, and general operations of state PWS programs. The primacy agency reporting requirements remain unchanged and would apply to PFAS as with any other regulated contaminant. The water system reporting requirements are mentioned in § 141.904 and cross-reference the reporting timeframes and provisions in § 141.31.

2. Summary of Major Public Comments and EPA Responses

A few commenters recommended that the EPA provide Data Entry Instructions within six months of the promulgation of the rule to allow primacy agencies, particularly those that do not use SDWIS State, to implement their data systems for reporting to the EPA, prepare their PWS, and train staff. The EPA acknowledges this comment and will work to develop Data Entry Instructions as soon as possible. One commentor recommended that the EPA provide separate tracking of reporting and monitoring violations. The EPA acknowledges this comment and will consider this as data reporting tools are developed. A couple of commentors recommended that the reporting and recordkeeping requirements for compliance within the rule should provide an option for not requiring the RAA to be reported by the laboratories if the primacy agency performs the RAA calculations for the water system. In addition, one commenter requested that the primacy agency calculate the RAA, and another commentor inquired whether the EPA intended to allow the water systems not to perform the RAA calculations if the primacy agency performs the RAA calculations. The EPA disagrees with these comments. To ensure that the water system has immediate knowledge of their compliance status, the final rule requires that water systems calculate the RAA and report this to the primacy agency. Primacy agencies or laboratories may also calculate the RAA, to confirm the results of the water system, but it is not a required reporting element under this regulation. Lastly another commentor suggested that utilities be required to report the occurrence and concentration of other PFAS listed in the method (preferably 533) to facilitate data collection and to better inform water treatment objectives. The EPA notes that many water systems are currently collecting samples and reporting monitoring data for 29 PFAS that can be measured with EPA Methods 533 and 537.1 under UCMR 5 where EPA has the regulatory authority.

3. Final Rule

The reporting requirements for primacy agencies under 40 CFR 142.15 remain unchanged and apply to PFAS as with any other regulated contaminant. The EPA intends to develop and provide access to Data Entry Instructions within one year after rule publication. The EPA will follow the usual protocol of engaging with a State-EPA workgroup for drafting the Data Entry Instructions. In this process, the EPA will consider the use of separate monitoring and reporting violation codes, like is used for the Revised Total Coliform Rule (RTCR). In this final regulation, the cross-reference to the water system reporting timeframes and provisions in § 141.31 at the start of § 141.904 is retained, and, at 40 CFR 141.904(b), table 2, the EPA requires water systems to report PFAS RAAs to their primacy agency. As a general process, the laboratory will conduct the analysis of the sample and the system will use the result to calculate their RAA; the RAA calculation may subsequently be completed by the primacy agency as a compliance check. The EPA does recognize that state laboratories often directly report results to the state as allowed in 40 CFR 141.31(c) and that electronic reporting tools, such as the Compliance Monitoring Data Portal (CMDP), may be used by systems to comply with this reporting requirement.

D. Exemptions and Extensions

1. Proposal

Pursuant to SDWA section 1412(b)(10), the EPA proposed that all systems must comply with the NPDWR three years after rule promulgation. The EPA's proposal acknowledged that a primacy agency or the EPA may grant an extension of up to two additional years to comply with an NPDWR's MCL(s) if the primacy agency or the EPA determines an individual system needs additional time for capital improvements. The EPA stated that ''[a]t this time, the EPA does not intend to provide a two-year extension nationwide.'' 88 FR 18689. The proposal also discussed how a state which has primary enforcement responsibility may exempt any individual system facing compelling factors, such as economic factors, additional time to comply with any requirement respecting an MCL of any applicable NPDWR under SDWA section 1416 (USEPA, 2023f).

2. Summary of Major Public Comments and EPA Responses

SDWA section 1412(b)(10) requires that a ''NPDWR shall take effect ''3 years after the date on which the regulation is promulgated unless the administrator determines that an earlier date is practicable.'' Section 1412(b)(2) also authorizes ''the Administrator, or a State (in the case of an individual system), may allow up 2 additional years to *comply with a maximum contaminant level* . . . if the Administrator or the State . . . determines that additional time is necessary for capital improvements'' (emphasis added). Congress intended the extension under this provision to allow for a total of five years to comply with the MCL. Thus, if the EPA provides a two-year extension of the MCL compliance deadline for all systems based on the need for capital improvements, a state cannot provide an additional two-year extension under section 1412(b)(10) for capital improvements but may grant exemptions under section 1416 consistent with applicable requirements.

Many commenters, including utilities and state primacy agencies, expressed difficulty in meeting the three-year compliance deadline. Commenters expressed that it will be very challenging to both conduct initial monitoring and take actions (*e.g.,* installing treatment) to comply with the MCL within three years. Many of these commenters shared their on-the-ground experience in managing facilities that required capital improvements and provided evidence that additional time is needed to procure, design, pilot, permit, and ultimately construct treatment systems. Additionally, several commenters provided evidence of on-going labor and workforce challenges as well as recent experience with supply chain difficulties to obtain materials necessary to design and construct treatment facilities, which many attributed as a direct or indirect result of the COVID-pandemic residual impacts (AWWA, 2023).

The agency has evaluated the data and information shared by commenters regarding their experience with the time it takes to implement capital improvement projects. The EPA estimates that approximately 4,100–6,700 systems will be impacted by the MCLs in this final rule. Based on the EPA's initial compliance forecast, the agency anticipates that many of these systems will be installing advanced treatment technologies to meet the final PFAS standards (for additional discussion on the compliance forecast, please see section XII). The treatment technologies listed as BAT for the final rule include GAC, ion exchange resins, and centralized RO/NF (please see section X for more information). To ensure cost effective compliance with the PFAS MCLs, systems often need to evaluate their treatment technology options as a first step. Several commenters have noted that this planning step may include pilot studies with potential treatment systems, or it may be limited to an evaluation of the raw water characteristics. Further, some commenters have submitted data and project management plans for systems choosing to conduct pilot testing, indicating that it may take a year or more to contract with vendors and to perform pilot testing. Once the planning step is completed, systems must design and construct the treatment systems. Several commenters submitted information to the EPA indicating that the design and permitting of the treatment systems can take an additional year or longer, and construction of the treatment system can take another year or longer. Because systems will also need time to obtain funding, obtain local government approval of the project, or acquire the land necessary to construct these technologies, many commenters contend that systems will need additional time beyond the three-year effective date to comply with the MCLs.

While the EPA stated in the proposed rule that the agency did not intend to provide a two-year extension nationwide necessary for capital improvements, the EPA finds that the evidence submitted by commenters strongly supports that a significant number of systems covered by this rule will need two additional years to make capital improvements to meet the MCL. Specifically, the EPA reviewed data from applicants seeking DWSRF funding for capital improvement projects (*e.g.,* installation of advanced treatment technologies such as GAC or IX) and confirmed that these projects, on average, take about three or more years to complete (which excludes the time and activities that may occur to ensure these capital improvement projects are implemented successfully, such as the time it may take to secure funding or to conduct pilot testing). This evidence along with the breadth of practicable experience shared by utilities and primacy agencies demonstrate that additional time is necessary for a significant number of system sizes and types located throughout the country to make capital improvements. Additionally, the EPA notes that the number of systems estimated to be impacted by the MCLs are greater than what the agency anticipated in the proposal (*i.e.,* an increase from 3,400–6,300 systems to 4,100–6,700 systems nationally). This increase provides further evidence that a capital improvement extension is warranted as the agency expects that many of these systems will be installing advanced treatment technologies to meet the final PFAS standards. The agency also agrees with commenters that on-going labor and workforce challenges exist and can limit the ability to design, construct and operate treatment facilities. These workforce challenges facing water utilities and other sector organizations support the need for a capital improvement extension as a sufficient availability of qualified personnel is necessary to implement and sustain capital improvement projects. These issues may be attributed as a direct or indirect result of the recent COVID–19 pandemic and are clearly documented in data submitted to the agency as part of the public comment process (AWWA, 2023). Based upon these considerations, the EPA determined, in accordance with section 1412(b)(10) of SDWA, that the compliance date for the PFAS MCLs, regardless of system size, will be 5 years from the date of promulgation of the standard.

Some commenters recommend the EPA to follow a staggered implementation timeframe similar to what was done in some previous NPDWRs where compliance deadlines were staggered based on system size (USEPA, 2001; USEPA, 2006a). In these prior examples, larger systems typically conducted their monitoring and implemented the MCL first, followed by smaller systems. Upon consideration of information submitted by commenters, particularly issues related to supply chain complications that are directly or indirectly related to the COVID–19 pandemic residual challenges, the EPA has determined that a significant number of systems subject to the rule, including large systems, will require two additional years to complete the capital improvements necessary to comply with the MCLs for PFAS regulated under this action. For this reason, the EPA disagrees with commenters that staggered implementation based on system size is warranted for this rule. While large systems may have greater resources to implement capital improvements (*e.g.,* engineering and construction management staff to manage the projects), they still require time to design, pilot, permit, and construct treatment facilities.

Some commenters note that it will be challenging for systems to conduct their initial monitoring and install treatment within three years, particularly for those systems not conducting UCMR 5 monitoring that is ongoing until 2026. The EPA notes that the agency is finalizing a flexibility for systems to use previously acquired monitoring data from UCMR 5 or an equivalent state-led monitoring program for their initial monitoring which is intended to alleviate the burden placed on water systems in collecting additional data (see section VIII of this preamble for additional information on monitoring). While the agency agrees that systems need an additional two years to make capital improvements, the EPA finds that it is practicable for most systems to complete their initial monitoring within three years because all systems serving greater than 3,300 people will have appropriate monitoring data from UCMR 5. Many systems smaller than 3,300 people will also have appropriate monitoring data from state-led monitoring programs that may be eligible to meet the rule's initial monitoring requirements, and some will have UCMR 5 or other data. If systems find elevated levels of PFAS, these systems have an additional two years to comply with the MCL. If a system does not have eligible previously collected monitoring data and are concerned about insufficient time to install capital improvements, the EPA encourages these facilities to collect monitoring data as soon as possible after rule promulgation, allowing them the bulk of the five-year period to plan for and install any capital improvements if necessary.

Some commenters point to concerns regarding laboratory capability and capacity in supporting the proposed three-year compliance timeline. Additionally, a couple of commenters noted that if additional time were allowed, water systems that are close to the MCL may have time to identify and address sources of PFAS in their watersheds rather than investing resources on treatment initially. Finally, a couple of commenters recommend the EPA consider implementation flexibilities for small and rural water systems and suggest that these types of utilities may not have staff capacity nor expertise to compete for funding to implement the rule. The EPA notes that these issues are not directly related to capital improvements and thus were not the basis for the EPA's decision to extend the compliance date for the PFAS MCLs. Although the EPA disagrees with assertions about insufficient laboratory capacity and capability at this time to support implementation of the NPDWR, to the extent there are initial implementation issues just after promulgation, extending the compliance date will also provide ancillary benefits toward addressing any such laboratory capability and capacity issues and may provide opportunities for systems who are close to exceeding the MCLs to investigate sources of contamination. Additionally, the extended compliance deadline may give smaller and rural water utilities more time to apply for funding under BIL (please see section II of this preamble above for a discussion on BIL). Further, other assistance programs such as the Environmental Justice Thriving Communities Technical Assistance Centers may provide additional fundamental training and capacity building activities for underserved and overburdened communities toward navigating Federal grant applications and managing funding opportunities.

The EPA requested comment as to whether there are specific conditions, in addition to the statutory conditions, that should be mandated for systems to be eligible for exemptions from the PFAS NPDWR under SDWA section 1416. Several commenters requested the EPA provide additional guidance to primacy Agencies on when exemptions are appropriate under SDWA section 1416 similar to what was done for the final Arsenic NPDWR (USEPA, 2002c). The EPA is not issuing additional guidance around implementation of SDWA section 1416 at this time but may consider it in the future. The EPA notes primacy agencies who have adopted the 1998 *Variance and Exemptions Regulation* (USEPA, 1998c) may choose to grant exemptions consistent with the requirements under this regulation to encourage systems facing compelling circumstances to come into compliance with the MCLs in an appropriate period of time.

3. Final Rule

Pursuant to SDWA section 1412(b)(10), the final PFAS NPDWR is effective June 25, 2024. The compliance date for the PFAS NPDWR, other than the MCLs, is April 26, 2027. As discussed above and upon consideration of information submitted by commenters, the EPA is exercising its authority under SDWA section 1412(b)(10) to implement a nationwide capital improvement extension to comply with the MCLs. All systems must comply with the MCLs by April 26, 2029. All systems must comply with other requirements of the NPDWR, including initial monitoring, by April 26, 2027.

Systems must comply with initial monitoring requirements within three years of rule promulgation and will be required to summarize PFAS monitoring results and applicable information beginning with CCRs delivered in 2027. As the MCL compliance date is set at five years from rule promulgation, systems must report MCL violations in the CCR, accompanied by the required health effects language and information about violations, starting in 2029. Monitoring and testing procedure violations require Tier 3 notification: systems must provide notice no later than one year after the system learns of the violation. Systems must repeat the notice annually for as long as the violation persists. Systems must comply with initial monitoring requirements within three years of rule promulgation and systems must provide Tier 3 notification for monitoring and testing procedure violations starting in 2027. As the MCL compliance date is set at five years from rule promulgation, systems must provide Tier 2 notification for MCL violations, starting in 2029. For more information on SDWA Right-to-Know requirements, please see section IX of this preamble above.

The agency notes that SDWA section 1416(a) and (b)(2)(C) describe how the EPA or states may also grant an exemption for systems meeting specified criteria that provides an additional period for compliance. PWSs that meet the minimum criteria outlined in the SDWA may be eligible for an exemption from the MCLs for up to three years. For smaller water systems (≤3,300 population), exemptions can provide up to six additional years to achieve compliance with the MCLs. States exercising primacy enforcement responsibility must have adopted the 1998 *Variance and Exemption Regulation* (USEPA, 1998c) for water systems in those jurisdictions to be eligible for an exemption.

XII. Health Risk Reduction and Cost Analysis

This section summarizes the final rule Health Risk Reduction and Cost Analysis (HRRCA) supporting document (USEPA, 2024g) for the per- and polyfluoroalkyl substances (PFAS) National Primary Drinking Water Regulation (NPDWR), which is prepared in compliance with section 1412(b)(3)(C) of the Safe Drinking Water Act (SDWA) and under Executive Order (E.O.) 12866. Section 1412(b)(3)(C)(i) lists the analytical elements required in a HRRCA applicable to an NPDWR that includes a Maximum Contaminant Level (MCL). The prescribed HRRCA elements include:

(1) Quantifiable and nonquantifiable health risk reduction benefits;

(2) quantifiable and nonquantifiable health risk reduction benefits from reductions in co-occurring contaminants;

(3) quantifiable and nonquantifiable costs that are likely to occur solely as a result of compliance;

(4) incremental costs and benefits of each alternative MCL considered;

(5) effects of the contaminant on the general population and sensitive subpopulations including infants, children, pregnant women, the elderly, and individuals with a history of serious illness;

(6) any increased health risks that may occur as a result of compliance, including risks associated with co-occurring contaminants; and

(7) other relevant factors such as uncertainties in the analysis and factors with respect to the degree and nature of the risk.

Based on this analysis, the Administrator confirms the finding made at proposal under section 1412(b)(4)(C) of SDWA that the quantified and nonquantifiable benefits of the MCLs justify the costs. The complete HRRCA for the final NPDWR is commonly referred to as the ''Economic Analysis'' (or EA) in this final rule and can be found in the docket at USEPA (2024g).

Because this NPDWR is promulgated in 2024 and provides a 2-year nationwide extension of the date for MCL compliance, the EA assumes that capital improvements (*i.e.,* installation of treatment technologies) for systems taking action under the rule will be completed by five years from the date promulgated, or in 2029. All other requirements, including initial monitoring, are assumed to be completed within three years of rule promulgation, or by 2027. Based on an assumed mean human lifespan of 80 years, the Environmental Protection Agency (EPA) evaluates costs and benefits under the final rule through the year 2105.

The EPA selected this period of analysis to capture health effects from chronic illnesses that are typically experienced later in life (*i.e.,* cardiovascular disease [CVD] and cancer). Capital costs for installation of treatment technologies are spread over the useful life of the technologies. The EPA does not capture effects of compliance with the final rule after the end of the period of analysis. Costs and benefits discussed in this section are presented as annualized present values in 2022 dollars. The EPA determined the present value of these costs and benefits using a discount rate of 2 percent, which is the discount rate prescribed by the Office of Management and Budget (OMB; OMB, 2023). All future cost and benefit values are discounted back to the initial year of the analysis, 2024, providing the present value of the cost or benefit.

Estimates of PFAS occurrence used for cost-benefit modeling rely on a Bayesian hierarchical estimation model of national PFAS occurrence in drinking water (Cadwallader et al., 2022) discussed in section VI.E. of this preamble. The model was fitted using sample data from systems participating in PFAS sampling under the third Unregulated Contaminant Monitoring Rule (UCMR 3) and included all systems serving over 10,000 customers and a subset of 800 smaller systems. A best-fit model was selected using sample data to define occurrence and co-occurrence of perfluorooctanoic acid (PFOA), perfluorooctane sulfonic acid (PFOS), and perfluorohexane sulfonic acid (PFHxS [12]) in water systems stratified by system size and incorporating variations within and among systems. Sample data were derived from state-level datasets as well as from UCMR 3. For more information on the EPA's occurrence model, please see section VI.E. of this preamble.

In the EA, the EPA analyzes the costs and benefits of the final rule, which includes MCLs for PFOA and PFOS at 4.0 ng/L each and MCLs for PFHxS, perfluorononanoic acid (PFNA), and hexafluoropropylene oxide dimer acid (HFPO–DA) at 10 ng/L each and a unitless Hazard Index (HI) of 1 for any mixtures of PFHxS, PFNA, HFPO–DA, and PFBS. The EPA also analyzed the costs and benefits for several regulatory alternatives. The EPA analyzed the costs and benefits of setting individual MCLs for PFOA and PFOS at 4.0 ng/L, 5.0 ng/L, and 10.0 ng/L, referred to as regulatory alternative MCLs under option 1a, option 1b, and option 1c, respectively. The EPA assessed these regulatory alternative MCLs in the EA to understand the impact of less stringent PFOA and PFOS MCLs. Additionally, the EPA has separately estimated national level marginal costs associated with the individual MCL for PFHxS if this MCL were to be promulgated in the absence of the Hazard Index; see chapter 5.1.3 of the EA for details. The EPA has also estimated the marginal costs for the individual PFNA and HFPO–DA MCLs if there were no Hazard Index in the sensitivity analysis found in appendix N.4. The EPA notes that the costs for the individual PFHxS, PFNA, and HFPO–DA MCLs have been considered in this final rule.

Section A summarizes public comments received on the EA for the proposed rule and the EPA's responses to comments. Section B summarizes the entities which would be affected by the final rule and provides a list of key data sources used to develop the EPA's baseline water system characterization. Section C provides an overview of the cost-benefit model used to estimate the national costs and benefits of the final rule. Section D summarizes the methods the EPA used to estimate costs associated with the final rule. Section E summarizes the nonquantifiable costs of the final rule.[13] Section F summarizes the methods the EPA used to estimate quantified benefits associated with the final rule. Section G provides a summary of the nonquantifiable benefits associated with reductions in exposure to both PFOA and PFOS expected to result from the final rule. Section H provides a qualitative summary of benefits expected to result from the removal of PFAS included in the Hazard Index component of the final rule and additional co-removed PFAS contaminants. Section I of this preamble summarizes benefits expected to result from the co-removal of disinfection byproducts (DBPs). Section J provides a comparison of cost and benefit estimates. Section K summarizes and discusses key uncertainties in the cost and benefit analyses. Quantified costs and benefits for the final rule and regulatory alternative MCLs under options 1a-1c are summarized in section XII.J, specifically Tables 68–71. Tables 72–73 summarize the non-quantified costs and benefits and assess the potential impact of nonquantifiable costs and benefits on the overall cost and benefit estimates for the final rule.

A. Public Comment on the Economic Analysis for the Proposed Rule and EPA Response

1. Methods for Estimating Benefits

a. Methods for Estimating Benefits in the Proposed Rule

In the EA for the proposed rule, the EPA presented quantified and nonquantifiable health benefits expected from reductions in PFAS exposures. Quantified benefits are assessed as avoided cases of illness and deaths (or morbidity and mortality, respectively) associated with exposure to some of the regulated PFAS contaminants. The EPA provided a quantitative estimate of CVD, birth weight, and renal cell carcinoma (RCC) avoided morbidity and mortality associated with reductions in PFOA and PFOS consistent with the proposed rule. The EPA also developed a quantitative analysis for reductions in bladder cancer morbidity and mortality that stem from removal of DBP precursors as a function of PFAS treatment. Adverse human health outcomes associated with PFAS exposure that cannot be quantified and valued are assessed as nonquantifiable benefits.

The EPA qualitatively summarized potential health benefits associated with reduced exposure to PFAS other than PFOA and PFOS in drinking water. In the proposal, the EPA discussed non quantified benefits associated with health endpoints including developmental effects, cardiovascular effects, hepatic effects, immune effects,

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[12] The EPA notes that perfluoroheptanoic acid (PFHpA) is not included in the proposed or final PFAS NPDWR; however, it was included in the occurrence model because of its UCMR 3 occurrence data availability; please see *Cadwallader et al., 2022* for additional details.

[13] This section includes costs with generally greater uncertainty that the EPA assesses in quantified sensitivity analyses.

endocrine effects, metabolic effects, renal effects, reproductive effects, musculoskeletal effects, hematological effects, other non-cancer effects, and COVID–19.

b. Summary of Major Public Comments on Method for Estimating Benefits and EPA Responses

Overestimation of Quantified Benefits

The EPA received comments from industry groups and organizations representing water utilities about the EPA's methodology for estimating quantitative benefits associated with the NPDWR. While some commenters supported the EPA's analysis, a few commenters stated that the agency overestimated quantified benefits. These commenters asserted that the EPA overstated the benefits of the rule and that the HRRCA is flawed because the existing health evidence does not support the quantified benefits. The EPA disagrees with commenters that the existing evidence does not support the EPA's estimate of quantified benefits from avoided adverse health effects likely to occur as a result of treatment and that these benefits are overstated. Among other things, the EPA has used the best available science in three key respects: by (1) considering relevant peer-reviewed literature identified by performing systematic searches of the scientific literature or identified through public comment, (2) relying on peer-reviewed, published EPA human health risk assessment methodology (USEPA, 2022f), and (3) utilizing peer-reviewed methodologies to valuing and quantifying avoided adverse health outcomes. Specifically, the EPA identified the full range of expected human health outcomes, including quantified benefits associated with co-removal of co-occurring contaminants (*i.e.,* DBPs). This process was built upon multidisciplinary research, including hazard identification and dose-response analysis, exposure assessment, and economic valuation methods recommended by the EPA's *Guidelines for Preparing Economic Analyses* (USEPA, 2016e) and updated Circular A–4 Guidance (OMB, 2023) to enumerate all beneficial outcomes, identify beneficiaries, and determine human health endpoints that can be valued. The EPA notes that the benefits analysis contains uncertainties associated with the modeling inputs in each of the steps listed above. In accordance with OMB Circular A–4 guidance (OMB, 2023), the EPA characterizes sources of uncertainty in its quantitative benefits analysis and reports uncertainty bounds for benefits estimated for each health endpoint category modeled in the final rule. See Table 75 and also section 6.1 of the EA for the final rule (USEPA, 2024g) for the list of quantified sources of uncertainty in benefits estimates. The reported uncertainty bounds reflect the best available data on health effect-serum slope factors, baseline PFAS occurrence, population size and demographic composition, and the magnitude of PFAS concentration reductions. In addition, some model inputs did not have sufficient distributional data to be included in the quantitative uncertainty analysis, and there are also uncertainties that could not be assessed quantitatively. These sources of uncertainty are described in Table 62 and also in section 6.8 of the EA for the final rule (USEPA, 2024g). Although some imprecision in the estimated benefits may be expected due to the lack of perfect information, the EPA has demonstrated, using the best science and data available, that there is sufficient health evidence to support the estimation of quantified benefit values and that these values are not systematic overestimates of the welfare improvements derived from implementation of the NPDWR.

Another commenter claimed that ''for the large majority of health endpoints discussed, the EPA has not provided a factual basis by which to conclude that such benefits are likely to occur when the EPA decreases the levels of PFAS in drinking water.'' The EPA disagrees with the commenter's assertion that the agency has not provided a factual basis for the benefits that are likely to occur as a result of the rule, which is amply supported in the HRRCA by the best available peer-reviewed science, consistent with SDWA section 1412(b)(3). Moreover, the commenter did not provide any additional or contrary factual information for the EPA to consider.

One commenter stated that the EPA did not provide data to support the analysis of benefits predicted from the implementation of the Hazard Index MCL. The EPA disagrees with commenter that the EPA did not provide evidence to support Hazard Index MCL benefits. In section XII of the preamble and in section 6.2 of the EA (USEPA, 2024g), the EPA qualitatively summarized and considered the potential health benefits resulting from reduced exposure to PFAS other than PFOA and PFOS in drinking water. These qualitative potential health benefits are based on summaries of a significant body of peer reviewed science. As summarized in the EA, the qualitatively discussed health effects of the Hazard Index PFAS are considerable; reducing human exposure to the Hazard Index PFAS is expected to reduce the incidence of multiple adverse health impacts. The qualitative benefits discussion of the impacts of the four PFAS which are regulated through the Hazard Index, as well as their co-occurrence in source waters containing PFOA and/or PFOS and additive health concerns, supports the EPA's decision to regulate them through the Hazard Index in this rulemaking.

Additionally, the EPA evaluated the impacts of PFNA (one of the Hazard Index PFAS) on birthweight in quantitative sensitivity analyses (USEPA, 2024e). The EPA notes that new evidence since the release of the current, best available peer reviewed scientific assessment for PFNA (ATSDR, 2021) provides further justification for the EPA's analysis of potential economic benefits of PFNA exposure reduction and avoided birthweight effects. Specifically, this new evidence confirms that in instances where PFNA is present, the national quantified benefits may be underestimated; however, birth weight benefits are considered quantitatively as part of this EA in the sensitivity analysis and support the EPA's decision to regulate PFNA.

The EPA received a number of comments on the quantitative analysis for CVD risk reduction. These commenters disagree with the EPA's assessment that cardiovascular benefits are likely to occur as a result of PFOA and PFOS exposure reduction. One commenter stated that the associations with total cholesterol (TC) are not biologically significant and criticized the EPA's use of linear models in the CVD meta-analysis, stating that this approach biases the analysis by excluding higher-quality studies. The EPA disagrees with the commenter's statement that associations between PFOA/PFOS and TC are not biologically significant. Such serum lipid changes may or may not result in a concentration considered clinically elevated in a particular individual; however, given the distribution of individual concentrations within the population, small changes in average serum lipid concentrations can result in substantial adverse health effects at the population level (Gilbert and Weiss, 2006). The EPA disagrees with the commenter's suggestions that linear assumptions are inappropriate for use in this context. The EPA presents the exposure-response estimates evaluated considering all studies, studies with linear models only, and a variety of sensitivity analyses in appendix F of the

EA (Tables F–2 and F–3, USEPA, 2024e). Meta-analyses of studies reporting linear associations had statistically significant relationships. These relationships are supported by the EPA's review of epidemiological studies showing positive associations between PFOA/PFOS and TC. The EPA used data from peer-reviewed studies, and the assumption of linear exposure-response function to explain associations between PFAS and serum lipids such as TC which are supported by data from numerous studies, including those used in the meta-analysis. Other studies have explored log-linear or linear-log relationships between PFAS and serum lipids, while acknowledging only ''slight improvements'' in model fit, especially for serum lipids with least skewed distributions (Steenland et al., 2009).

A couple of commenters stated that the downward trend in decreasing total and low-density lipid cholesterol since the 1970s coupled with the decreasing PFOA and PFOS serum levels suggests that there is a substantial likelihood that the proposed MCLs for PFOA and PFOS are unlikely to result in benefits as great as those reported in the proposal. The EPA disagrees with these comments asserting that decreasing trends in cholesterol levels over time indicate that PFAS exposure is unlikely to contribute to a measurable increase in CVD risk. The EPA relied on recent National Health and Nutrition Examination Study (NHANES) data (2011–2016) to inform baseline cholesterol and blood pressure conditions in the population evaluated under the proposed rule. These data reflect the current population and do not reflect cholesterol conditions in the population between 1970 and 2010. Therefore, the CVD benefits analysis examines how the probability of the current population might benefit from reduced incidence of hard CVD events.[14]

The EPA received a comment stating that the benefits associated with high-density lipoprotein cholesterol (HDLC, often referred to as the 'good cholesterol') changes are not likely to accrue because the evidence of the relationship between PFAS and the health outcome is not conclusive, and that this endpoint should not have been quantified. The EPA disagrees; although the evidence of a relationship between PFAS exposure and HDLC is not conclusive, the SAB recommended that the EPA evaluate how the inclusion of HDLC effects would influence results. Thus, the EPA evaluated how benefits results are affected by the inclusion of HDLC effects in a sensitivity analysis presented in appendix K of the EA for the proposed (USEPA, 2023f) and final rule (USEPA, 2024e). Additionally, the same commenter and one other commenter challenged the EPA's quantification of PFOS and blood pressure, stating that the EPA's finding that PFOS might have ''the potential'' to affect blood pressure does not meet the SDWA standard for inclusion in a benefits analysis and that the ''rationale for including changes in BP in relation to PFOS is not clear.'' Another comment identified a study that utilized NHANES data and ''did not observe an association'' between PFOA and blood pressure. Finally, another commenter mentioned that ''neither the ATSDR nor the National Academy of Sciences (NAS) have found an association between PFOA/PFAS and increased blood pressure.'' While the EPA is aware of this previous work, in the EPA's own, more recent assessment, the strength of the evidence is determined both by the number but also the quality of studies investigating the relationship. One high confidence study conducted using U.S. general population data from NHANES showed a relationship between PFOS exposure and systolic blood pressure in humans (Liao et al., 2020). In addition, several *medium* and *low* confidence studies provided evidence for an association between PFOS and blood pressure and/or hypertension (Mitro et al., 2020; Bao et al., 2017; Mi et al., 2020; Liu et al., 2018). Because blood pressure is an important component of the Atherosclerotic Cardiovascular Disease (ASCVD) model used to estimate hard CVD event risk, and because epidemiology reports show consistent evidence of an association between PFOS and blood pressure in general adult populations (*i.e.,* the populations evaluated using the ASCVD model), the EPA included the relationship between PFOS exposure and blood pressure in the analysis. The EPA further notes that the Science Advisory Board recommended modeling the impacts of changes in all ASCVD model predictors (including blood pressure and HDLC) for which there is evidence of a likely causal relationship (USEPA, 2022i).

A few commenters questioned the evidence or stated that the evidence supporting an association between exposure to PFOA and PFOS and CVD is insufficient. The EPA disagrees with these comments. The agency's approach to estimating reductions in CVD risk was reviewed and supported by SAB panelists (USEPA, 2022i). Numerous studies have shown consistent associations between PFOA/PFOS exposure and changes in TC and blood pressure which are biomarkers for CVD risk. TC and blood pressure are well-established CVD risk biomarkers, are clearly associated with CVD events, and are important inputs to the ASCVD model that the EPA used to estimate CVD outcomes.

The EPA received public comments on the benefits analysis for developmental effects. A few commenters claimed that the studies used for developmental modeling did not provide sufficient evidence of an association between PFOA and PFOS exposure and stated that the studies which the EPA used to model the developmental effects relationship did not consider confounders including pregnancy hemodynamics and other chemical and non-chemical stressors, including other PFAS. One commenter stated that the EPA's findings are inconsistent with other regulatory agency findings that small decreases in birth weight are associated with maternal exposure to PFOA and PFOS but not increased risk of low birth weight. Other commenters stated that the EPA did not address these concerns and inappropriately used these studies to support quantitative analysis, and one commenter stated that because of the shortcomings of the studies used and the modeling uncertainties, peer review of the developmental effects modeling should be completed. Although there are some uncertainties in the developmental epidemiological effects data (*e.g.,* differences seen across biomarker sample timing), the EPA disagrees with these comments: the developmental benefits analysis is supported by a wide body of peer reviewed science (Verner et al., 2015; Negri et al., 2017; ATSDR, 2021; Waterfield et al., 2020; USEPA, 2016c; USEPA, 2016d; USEPA, 2024c; USEPA, 2024d). Specifically, birth weight was determined to be a critical effect based on findings in the EPA's health assessments (see USEPA, 2024c; USEPA, 2024d), and low birth weight is linked to a number of health effects that may be a source of economic burden to society in the form of medical costs, infant mortality, parental and caregiver costs, labor market productivity loss, and education costs.

Discussion regarding the selection of decreased birth weight as a critical effect, including the selection of specific studies for candidate RfD derivation and the evidence supporting associations between PFOA or PFOS and

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[14] Hard CVD events include fatal and non-fatal myocardial infarction (*i.e.,* heart attack), fatal and non-fatal stroke, and other coronary heart disease mortality.

developmental effects, is available in sections 3.4.4 and 4.1 of the final toxicity assessments for PFOA and PFOS (USEPA, 2024c; USEPA, 2024d). In estimating benefits of reducing PFOA and PFOS in drinking water, the agency selected results from Steenland et al. (2018) as the birth weight exposure-response function for PFOA and results from Dzierlenga et al. (2020) as the birth weight exposure-response function for PFOS. The agency chose the results from these studies because they include the most recent meta-analyses on PFOA- and PFOS-birth weight relationships, and they included a large number of studies, including multiple studies with first trimester samples (seven studies in Steenland et al., 2018 and eight studies in Dzierlenga et al., 2020). To provide insights into the potential effects of sample timing and pregnancy hemodynamics, the EPA also performed a sensitivity analysis considering only first trimester estimates from Steenland et al (2018) for PFOA and Dzierlenga et al. (2020) for PFOS in section K.4 of the EA appendices (USEPA, 2024e). While reports prior to 2019 found ''plausible'' or ''suggestive'' (USEPA, 2016d; ATSDR, 2018) evidence of relationships between PFOA and PFOS and developmental outcomes, the EPA's assessment found clear evidence of an association for PFOA and PFOS in both toxicological and epidemiological studies (USEPA, 2024h; USEPA, 2024i). The agency further disagrees with the commenter's statement that further peer review is needed, as the EPA relies extensively on peer-reviewed studies in its developmental benefits model. Furthermore, the EPA characterizes the uncertainty in the PFOA and PFOS exposure-response functions as described in appendix L of the EA (USEPA, 2024e). In short, the benefits analysis for developmental effects relies on a wide body of the best available, peer-reviewed science, and the epidemiological evidence provides a reliable basis for quantifying the risks of low birth weight.

A different commenter claimed that the EPA relied on equivocal epidemiological evidence to estimate developmental benefits, stating that the RfDs calculated from animal studies in the EPA's health assessment documents for PFOA and PFOS are significantly higher than those based on human studies used for benefits analysis and that the animal studies represent a more appropriate estimate of the risk of PFOA and PFOS exposure. The EPA disagrees with the commenter that the analysis relies on equivocal epidemiological evidence to estimate benefits. The systematic literature review and assessment conducted by the EPA, the most comprehensive evaluation of the current literature to date, concluded that there is moderate evidence for developmental effects based on consistent adverse effects for fetal growth restriction including birthweight measures which are the most accurate endpoint (USEPA, 2024c; USEPA, 2024d). One commenter raised concerns about the EPA's reliance on the study (Steenland et al., 2018) that the EPA uses to model PFOA dose response for benefits analysis, stating that the EPA's benefits analysis for PFOA and developmental effects is not supported by the underlying publication. The same commenter questioned the EPA's reliance on the study that is used to model PFOS dose response for benefits analysis (Dzierlenga et al., 2020), stating that the study found that there was no evidence of a relationship at the beginning of pregnancy. The commenter contended that the meta-analysis was not peer reviewed and thus the validity of the EPA's methods should be questioned. The EPA disagrees with the commenter's criticism of the studies used to assess dose response in developmental benefits analysis. The selected meta-analyses on the relationship between PFOA/PFOS exposure and birth weight produced statistically significant results, are based on recent data, and include a large number of studies in each meta-analysis.

One commenter stated that given the discussion about changes over time in infant mortality, a dataset containing only two years of data is insufficient to build infant mortality regression models. The EPA disagrees that two years of data is insufficient to build regression models relating infant birth weight to infant mortality. The EPA's regression analysis improves upon earlier analyses relating birth weight to infant mortality (Almond et al., 2005; Ma and Finch, 2010) by evaluating two years of recent data. Sample sizes among the Centers for Disease Control and Prevention (CDC) National Center for Health Statistics (NCHS) linked birth/infant death data per year are large (n = approximately 3.8 million infants) and contribute to the overall statistical significance of regression results. As described in appendix E of the EA (section E.2, USEPA, 2024e), there has been a notable decline in U.S. infant mortality rates since the analyses reported in Ma and Finch (2010) and Almond et al. (2005). Using recent data from two CDC NCHS linked birth/infant death data cohorts results is a more accurate and conservative characterization of recent infant mortality trends than if the EPA had included older CDC NCHS data.

The EPA received comments on the benefits analysis for RCC. Two commenters expressed concerns with the EPA's use of Shearer et al. (2021) to estimate RCC risk in benefits analysis and claimed flaws in the study related to outliers in the RCC group and inconsistent evidence of an association across epidemiological studies. One commenter stated that given what they perceive as SAB concerns and uncertainties in the modeling, further peer review is warranted. The EPA disagrees with the comments critical of the agency's use of information from the Shearer et al (2021) study for purposes of PFOA health assessment and benefits analysis. As noted in section 3.5.1 of the Final Toxicity Assessment for PFOA (USEPA, 2024c), the EPA determined that Shearer et al. (2021) is a *medium* confidence study after conducting study quality evaluation consistent with the *ORD Staff Handbook for Developing IRIS Assessments* (USEPA, 2022f). The biomonitoring measures of PFOA levels in Shearer et al. (2021) were reliable measures of PFOA exposure due to the chemical's well-established long half-life. The commenters failed to acknowledge multiple studies further supporting a positive association between PFOA exposure and RCC risk (Bartell and Vieira, 2021; Vieira et al., 2013; Steenland et al., 2022). Critically, the SAB PFAS Review Panel supported the *Likely to be Carcinogenic to Humans* designation for PFOA in its final report (USEPA, 2022i). Shearer et al (2021) has been sufficiently peer reviewed and it represents the best available science for purposes of health and benefits assessment in the PFAS NPDWR.

The EPA received comments on uncertainties associated with bladder cancer reductions. One commenter incorrectly stated that the ''EPA does not recognize the uncertainty that there is not always direct correlation between THM4 levels and TOC in all public water systems''. In response, the EPA notes that the THM concentrations in this co-removal analysis were not calculated based on TOC reduction. TOC was used to bin systems in the universe of PWSs using the fourth Six-Year Review (SYR4) database and PFAS occurrence model with the THM4 reduction calculated from the formation potential experiments before and after GAC treatment in the DBP Information Collection Rule Treatment Study Database. This dataset reflects the current best available data to determine THM4 reduction based on TOC removal using GAC treatment. Another commenter stated that the causal link of DBPs and bladder cancer has not been established. The EPA notes that an extensive body of epidemiological studies have shown that increased exposure to chlorinated DBPs is associated with higher risk of bladder cancer and other adverse health outcomes (Cantor et al., 1998; Freeman et al., 2017). Weisman et al. (2022) found that approximately 8,000 of the 79,000 annual bladder cancer cases in the U.S. were potentially attributable to chlorinated DBPs in drinking water systems. While research has not established a causal link between THM4 and bladder cancer, there is strong evidence that there is a correlation between THM4 and bladder cancer.

One commenter stated that the DBP co-removal benefit analysis did not meet the standards required by SDWA for estimating benefits since it was not reviewed by the SAB. The commenter is incorrect. SDWA 1412(e) directs the EPA to request comments from the SAB prior to proposing an MCLG and NPDWR. The EPA sought and received comment from the SAB prior to proposing this NPDWR (see USEPA, 2022i). The statute does not dictate the precise level of scientific questions for which the EPA must seek comments from the SAB. The EPA sought SAB comment on the four most significant areas that informed derivation of the MCLGs for all six PFAS regulated by this action and for other parts of the benefits analysis that informed the overall development of the NPDWR. The EPA did seek additional peer review of its DBP co-removal benefit analysis prior to its inclusion in the EA for which it received overwhelmingly favorable comments from reviewers (see USEPA, 2023m). Furthermore, this rule is based on the EPA's consideration of a wide body of existing peer-reviewed science on this subject (*e.g.,* Regli et al., 2015; Weisman et al., 2022). In short, the EPA has used peer reviewed science and sought further peer review to support its DBP co-removal analysis, and as part of the supporting material for the rule proposal, the EPA included the comments from the expert peer reviewers as well as how each comment was addressed or the rationale for why it was not changed. Please see *Response to Letter of Peer Review for DBP Co-benefits* (USEPA, 2023m) for discussion of that peer review and the EPA's responses to peer reviewed comments.

Another commenter claimed that the EPA improperly quantified benefits of co-removed substances rather than co-occurring substances. The EPA disagrees with these assertions since the analysis of DBP co-removal is focused on co-occurring contaminants. As demonstrated elsewhere in the record for this action, PFAS commonly co-occur with each other. Additionally, in waters where disinfection is required, TOC (*i.e.,* a DBP precursor) and PFAS may co-occur. The DBP co-removal benefits analysis relied on DBP formation potential experiments that highlighted the changes to TOC with and without GAC treatment. Furthermore, as discussed above, the methodology to estimate THM4 reductions was externally peer reviewed by three experts in GAC treatment for PFAS removal and DBP formation potential.

A few commenters stated that the EPA already had initiatives to reduce THMs in drinking water and suggested that reduction of bladder cancer cases is better addressed through existing DBP rules. While the EPA agrees that there are existing DBP regulations to reduce DBP exposure and risks, this rule will provide additional health risk reduction benefits associated with enhanced DBP reduction. The EPA has considered those co-removal benefits as part of the EA. The EPA notes that it is required under the SDWA 1412(b)(3)(C)(i)(II) to assess quantifiable and nonquantifiable health risk reduction benefits for which there is a factual basis in the rulemaking record to conclude that such benefits are likely to occur from reductions in co-occurring contaminants that may be attributed solely to compliance with the MCL, excluding benefits resulting from compliance with other proposed or promulgated regulations. DBP reductions presented in the EPA's HRRCA are those that are anticipated to result solely from compliance with the PFAS MCLs. As required under the SDWA, any quantifiable and nonquantifiable benefits from future actions concerning DBPs in drinking water will be addressed at the time of those actions and are independent from benefits stemming as a result of the PFAS rulemaking. A couple of commenters supported the EPA's analysis of DBP benefits but recommended that the EPA also consider other co-removed contaminants. The EPA agrees with the commenters that multiple co-occurring contaminants will be removed as a result of this rule. Furthermore, the EPA acknowledges in the EA that additional co-removal benefits would be realized due to treatment for PFAS. With the exception of DBPs co-removed, the EPA has not quantified other co-removal benefits at this time because of data limitations, the agency included discussion of nonquantifiable benefits for multiple other PFAS and for other contaminants.

Nonquantifiable Benefits of PFAS Exposure Reduction

One commenter expressed that the EPA's characterization of benefits is inadequate and not supported by science. The commenter specifically discussed hepatic effects, endocrine effects, and musculoskeletal effects and asserted that the EPA's characterization is based on mixed findings and inconsistent evidence regarding PFAS exposures and specific health outcomes. The EPA disagrees with this comment, as the EPA has evaluated the best available peer reviewed science, as required under SDWA. The EPA did not quantify or monetize benefits where there are inadequate data. For hepatic effects, the EPA's toxicity assessments determined that there is moderate evidence supporting the association between exposure to PFOA/PFOS and hepatic toxicity in humans. However, the EPA did not quantify benefits for hepatic effects because although there will be benefits delivered by reducing PFOA and PFOS in drinking water, there is a lack of adequate data available to accurately quantify those benefits. Further information on health effects related to PFAS exposures is provided in the health assessments within the MCLG documents (USEPA, 2024c; USEPA, 2024d).

Conversely, some commenters expressed support for the quantification that the EPA has already performed, stated that the benefits of the rule are underestimated, and urged the EPA to quantify and monetize additional health endpoints, particularly mammary gland and lactational effects, immunotoxicity, and liver disease. These commenters also provided additional resources and information with the intention of the EPA using that information to update analyses regarding lactational effects, expand analyses to include immune effects, and adjust analyses to characterize hepatotoxicity as a quantifiable benefit, as opposed to a non-quantifiable one. Commenters also urged the EPA to quantify some of the benefit categories, even if monetization is not possible, and to highlight the magnitude of some of the qualitatively discussed benefits. The EPA agrees with these commenters that the quantified benefits of the rule are underestimated. Where appropriate, the EPA used medical cost information provided by the commenters to supplement qualitative discussion of adverse effects. Additionally, and based on these comments, the EPA considered information in the record and added additional quantified benefits analysis in the sensitivity analysis evaluating the reductions in liver cancer cases expected by reducing concentrations of PFAS. This additional analysis was confirmatory of the EPA's previous analysis and did not result in changes to the NPDWR's requirements.

Some commenters also provided recommendations regarding the inclusion of additional costs and benefits beyond health endpoints. These included the opportunity cost of time, environmental benefits, and psychosocial benefits that are expected to result from the rule. The opportunity cost of time was suggested to be incorporated into morbidity estimates, while the other benefits were suggested to be encapsulated in a qualitative summary.

In the EA document, the EPA describes that the cost of illness (COI)-based approach does not account for the pain and suffering associated with non-fatal CVD events. Based on the above comments, for quantified cancer endpoints (*i.e.,* RCC and bladder cancers), the EPA has included a new sensitivity analysis using willingness to pay values for risk reductions which can inform the direction of benefits when opportunity cost is included. This additional analysis was confirmatory of the EPA's previous analysis and did not result in changes to the NPDWR's requirements.

c. Final Rule Analysis

For the final rule, the EPA retained the quantitative benefits analyses from the proposal for developmental, CVD, and cancer endpoints as well as the bladder cancer benefits from DBP exposure reduction as a result of the rule. In response to comments described above, the agency identified new information on willingness to pay values for non-fatal cancer risk reductions and added additional sensitivity analyses for RCC and bladder cancer in appendix K to the final rule EA (USEPA, 2024e). In light of new epidemiological studies on PFOS exposure and liver cancer that strengthened the weight of evidence and supported the toxicological information that was identified in the proposed rule, and comments received requesting that the EPA monetize additional health endpoints, the EPA developed a sensitivity analysis assessing the liver cancer impacts in appendix O of the final rule EA (USEPA, 2024e). The EPA estimates that PFOS liver cancer benefits would add $4.79 million annually to the national benefits estimates. The EPA retained discussion of nonquantifiable benefits associated with PFAS exposure reduction from the proposed rule for the final rule EA.

2. Treatment Costs

a. Treatment Cost Estimates in the Proposal

The EPA estimated costs associated with engineering, installing, operating, and maintaining PFAS removal treatment technologies, including treatment media replacement, and spent media destruction or disposal, as well as nontreatment actions that some PWSs may take in lieu of treatment, such as constructing new wells in an uncontaminated aquifer or interconnecting with and purchasing water from a neighboring PWS. To evaluate the treatment costs to comply with the proposed PFAS NPDWR, the EPA used the agency's Work Breakdown Structure (WBS) models, a spreadsheet-based engineering models for individual treatment technologies, linked to a central database of component unit costs. The WBS models are extensively peer-reviewed engineering models for individual treatment technologies and discussed in section XII.D of this preamble. The EPA used PFAS occurrence outputs from a Bayesian hierarchical estimation model of national PFAS occurrence in drinking water (Cadwallader et al., 2022), to estimate the number of water systems exceeding the proposed MCLs, and therefore triggered into action to comply with the proposed MCLs.

b. Summary of Major Public Comments on Treatment Costs and EPA Responses

Many commenters state that the EPA has underestimated the treatment costs required to comply with the proposed MCLs. One commenter suggested that the EPA has not complied ''with its statutory requirements by conducting an analysis that fully captures these costs.'' The EPA disagrees with the few commenters that suggested the EPA has not met its requirements under SDWA, and the EPA emphasizes the agency has used the best available peer reviewed science to inform it cost estimates, including treatment costs, of the MCLs. Specific aspects of comments related to treatment costs and the EPA's response are discussed further in this section.

Many commenters cited rising costs in the drinking water sector and discussed the effects of inflation and the COVID–19 pandemic on the costs of labor, construction, and capital, among other materials related to compliance with the MCLs. These commenters emphasized the significant impacts felt from supply chain and workforce issues. The EPA recognizes these impacts, and as recommended by commenters, adjusted the cost estimates by escalating unit costs using indices including the Bureau of Labor Statistics producer price indices (USBLS, 2010). The EPA updated each unit cost using the change in the relevant price index from year 2020 to 2022. For example, the EPA applied the percent increase of the price of metal tanks and vessels (50 percent increase from 2020 to 2022) to the price of metal tanks and vessels in the WBS cost models. The EPA also collected new vendor price quotes for cost driver equipment components (*e.g.,* pressure vessels, treatment media) and made several other adjustments to WBS model assumptions, described further in this section. Taken together, these adjustments increased the system level capital cost estimates in the EPA's cost assessment by a percentage that varied depending on the system size and treatment technology. For small systems using GAC and IX, the increase ranged from approximately 40 percent to 110 percent. For medium systems, the increase was approximately 20 to 60 percent; for large systems, 10 to 40 percent. Additionally, while revising the SafeWater model to incorporate new information from public comments, the EPA identified and corrected a coding error related to the discounting of future operation and maintenance costs resulting in increased estimated annualized treatment costs. The result of these changes are increased cost estimates for the final rule.

Some commenters state that while BIL funding is available, it is not enough to cover the compliance costs of the rule. For example, one commenter noted that, ''[t]his amount of funding support, while crucial, will come nowhere near the cost to ratepayers that must be borne to implement necessary compliance actions for these MCLs.'' The EPA disagrees with the commenter that BIL funding will be nowhere near the cost'' necessary to implement compliance actions. The EPA estimates that the initial capital costs of the rule in undiscounted dollars is approximately $14.4 billion (see appendix P of the EA for more information). Given the BIL appropriations of $11.7 billion in DWSRF and an additional $5 billion for emerging contaminants, the EPA reasonably anticipates BIL funding is likely to be able support a substantial portion of the initial capital costs of the final rule. BIL funding appropriations began in the Federal Fiscal Year (FFY) 2022 and appropriations are anticipated to continue through FFY 2026.

Many commenters shared some information about the costs that they have incurred or estimated they would incur at a system level to install, operate, and maintain treatment to remove PFAS. Some system level cost information provided by commenters fell within the ranges of costs presented in the EPA's supporting documentation for the proposal and other information provided by commenters exceeded the EPA's system level cost ranges. The EPA does not dispute the commenters stated experience of costs to install, operate and maintain treatment to remove PFAS; however, many of these comments lacked supporting details. Many of the comments cited preliminary or conceptual estimates and did not specify the methods and assumptions used to develop the estimate. Furthermore, most comments did not include information to confirm that all of the reported or estimated costs were or would be directly associated with PFAS treatment, as opposed to other infrastructure improvements (*e.g.,* capacity expansion, administrative facilities, distribution system improvements) that happened to be completed as part of the same project. Most commenters also did not include information to confirm that key design and operating parameters (*e.g.,* empty bed contact time, media replacement frequency) would be similar to the typical values assumed in the EPA's estimates. To fully evaluate the commenters' reported or estimated costs in comparison to WBS model results, the EPA would need itemized line-item cost details and engineering design parameters. To inform the cost estimates of the proposed and final PFAS NPDWR, the EPA conducted an extensive review of the literature. The EPA has further validated the unit costs in the PFAS rule with equipment cost information from 2023 from a major supplier of treatment media. While the EPA recognizes there are likely site-specific instances where costs exceed the EPA's cost ranges, there are also likely site-specific instances where costs are less than the EPA's cost ranges, and this level of accuracy is appropriate for a national level analysis.

Other commenters compared state-level costs to the EPA's national level cost estimates, noting that the EPA's estimates appeared too low. Utilizing this permit data and project cost data submitted by water systems in applications to the DWSRF, one state estimated that total capital costs for installation of PFAS treatment to meet the EPA's proposed standards across the state could be as high as $1.065 billion. The EPA's EA analysis, however, presents national level cost estimates that are annualized over the period of analysis and are therefore not directly comparable to a single year estimate of capital costs.

A few commenters stated that the EPA incorrectly omitted the costs associated with performance monitoring, which commenters believe will be necessary because a water system needs to know how often it needs to replace its media. The EPA disagrees that large amounts of additional samples in performance monitoring will be required, and the commenter provided no data to support their assertion that this would be necessary. The EPA anticipates that many water systems will conduct a pilot test before implementing a full-scale treatment installation and that the operational results from the pilot test will be a sufficient indicator of performance; therefore, water systems should not have to collect large amounts of performance samples indefinitely during the full-scale operation of treatment technologies. The EPA includes the costs of pilot testing, and sampling during that time, in the treatment capital cost estimates. In response to public comments, the EPA increased the estimated length of the pilot study and the frequency of sampling during the pilot study. Additionally, the EPA added a full year of confirmation sampling after full-scale installation to the estimated pilot study costs. Taken together, these changes doubled to more than tripled the pilot study costs included in the EPA's estimates.

In response to public comments about residual management concerns for high pressure membrane technologies, the EPA has adjusted RO/NF's technology projection compliance forecast to zero percent in the EA for the final rule. Therefore, the EPA assumes that RO/NF will not generally be used solely for the purpose of complying with the final rule. For more information on public comments on residuals management and the EPA's response please see section X.

A few commenters stated that the EPA underestimated or insufficiently incorporated contingency in its cost estimates. For example, one commenter stated that the EPA's contingency assumptions in the proposal were ''. . . inconsistent with recommended best practices for cost estimators and [are] expected to be a major contributor to the EPA WBS' failure to accurately capture costs for PFAS treatment facility implementation.'' In response to these comments, the EPA changed its approach and incorporated contingency for all systems, not just high-cost systems. The EPA also increased the complexity factor applied to estimate contingency for systems using GAC. Taken together, these changes result in a contingency factor of 5 to 10 percent depending on total project cost at all cost levels for systems installing treatment. Additionally, the EPA includes a miscellaneous allowance of 10 percent. This allowance can be viewed as either as a form of contingency or a method to increase the level of project definition (thus reducing the amount of contingency required).

One commenter stated that the EPA underestimated the costs associated with interconnection.[15] This commenter stated that it was ''unrealistic to assume that booster pumps are unlikely to be necessary. Pressure loss associated with friction could be significant, especially for an interconnection that may span 10,000 feet or more,'' and recommended that the EPA include booster pumps in the cost estimate. Commenters also pointed out that ''. . . systems considering interconnections will need to thoroughly investigate this option and determine if it is both cost effective and appropriate given the water quality impacts.'' In response to these comments, the EPA made several changes to the assumptions used to estimate costs for interconnection in the WBS model for nontreatment options. The EPA agrees that booster pumps may be needed and added the costs of booster pumps designed to account for friction loss in interconnecting piping. The EPA also agreed that there are many considerations for water systems pursuing interconnections including elevated water age, nitrification, and DBPs, as pointed out by commenters, and therefore the EPA increased the complexity factor applied to estimate contingency for systems using nontreatment options. Taken together with the escalation to 2022 dollars, these changes increased the system level capital costs for interconnection by approximately 60 to 100 percent.

Many commenters cited and expressed agreement with the conclusions of a study conducted by Black & Veatch on behalf of the American Water Works Association (AWWA) (hereafter referred to as AWWA's B&V report) (AWWA, 2023). The EPA disagrees with many of the assumptions in AWWA's B&V report and the report's overall conclusions

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[15] Interconnection is when a system replaces their contaminated water source by purchasing water from another nearby system that is in compliance. Booster pumps can be needed when the pressure from the supplying system is lower than required at the purchasing system and also to overcome pressure losses due to friction in interconnecting piping.

about the estimated national costs of the PFAS NPDWR. Tables 24–26 detail some of the key assumptions related to (1) PWSs that exceed the MCL, (2) capital costs and (3) operation and maintenance costs that overestimate national treatment costs in AWWA's B&V report and the EPA's response to those assumptions and resulting estimates. In combination, all these factors result in an overestimate of treatment costs. For example, AWWA's B&V report Table 6–1 reports an average capital cost per EP for the smallest size category of $900,000. Using AWWA's B&V report's (overestimated) design flow calculations, the treatment system design flow at each EP would be approximately 0.062 million gallons per day (mgd). For comparison, Forrester (2019) reports capital equipment costs of approximately $300,000 for a 1 mgd GAC PFAS treatment system. Even after adding indirect capital and building costs, the $900,000 estimate appears substantially overestimated, given that it is for a treatment system designed for approximately 1/16th of the flow of the system in the Calgon Carbon estimate (Forrester, 2019). When AWWA's B&V report's EP level results are aggregated nationally to an overestimated number of systems treating for PFAS, the overestimates are compounded at the national level.

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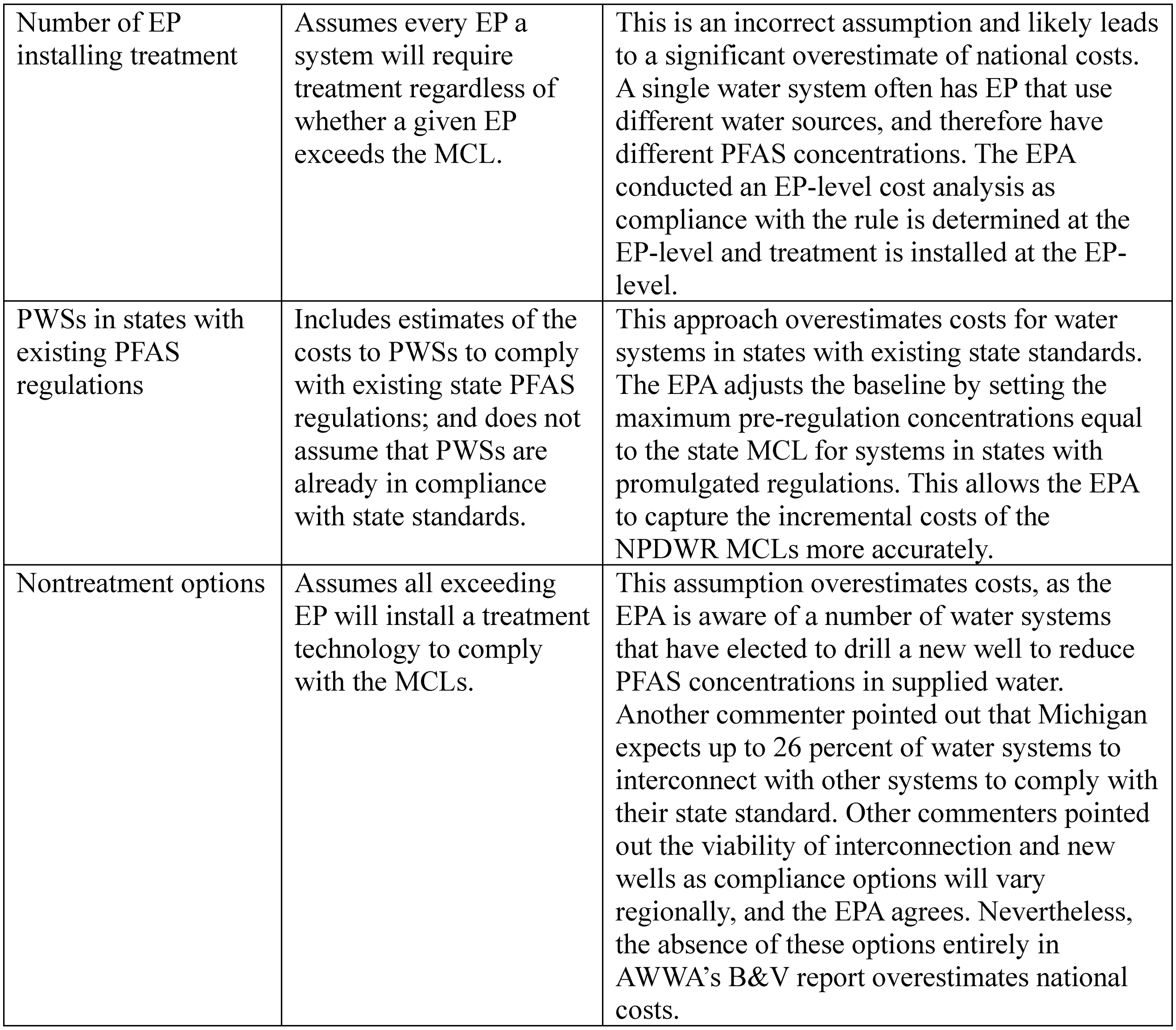
Table 24. EPA Response to assumptions about PWSs exceeding the MCLs in AWWA's B&V Report

Analytical Component	**AWWA's B&V report**	**EPA response**
PFAS occurrence estimates	Used an occurrence dataset comprised of UCMR 3 and information from state regulatory agencies. Estimates the following number of water systems will exceed 4.0 ng/L PFOA and/or PFOS: Serving 10,000 or less: 7,056 PWS (8,808 EP) Serving more than 10,000: 393 PWS (1,214 EP) Total PWSs: 7,449 PWSs (10,022 EP)	The dataset used is not appropriate for national extrapolation, for example, 90 percent of non-UCMR systems used in the report come from just 6 states. As a result, AWWA's B&V report likely overestimates the number of water systems exceeding the MCLs, particularly small water systems. After incorporating updated state monitoring data into its occurrence model, the EPA estimates the following number of water systems will exceed 4.0 ng/L of PFOA and/or PFOS (**mean** (5th – 95th) from chapter 4.4 of the EA): Serving 10,000 or less: **3,870** (2,795-5,097) PWS **5,115** (3,666-6,858) EP Serving more than 10,000: **1,266** (1,203-1,328) PWS **3,878** (3,701-4,056) EP Total PWSs: **5,136** (4,018-6,441) PWSs **8,993** (7,497-10,711) EP AWWA's B&V report did not specify what measures, if any, were taken to ensure the data was nationally representative and this may be one cause of their overestimation of water systems exceeding the MCLs. The EPA used QC measures to ensure that the data represented finished drinking water and that the set of systems used to inform the model was nationally representative. Additional state data that were available at systems that were part of this nationally representative set of systems were used to fit the model. For more information see section VI of this preamble.



Number of EP installing treatment	Assumes every EP a system will require treatment regardless of whether a given EP exceeds the MCL.	This is an incorrect assumption and likely leads to a significant overestimate of national costs. A single water system often has EP that use different water sources, and therefore have different PFAS concentrations. The EPA conducted an EP-level cost analysis as compliance with the rule is determined at the EP-level and treatment is installed at the EP-level.
PWSs in states with existing PFAS regulations	Includes estimates of the costs to PWSs to comply with existing state PFAS regulations; and does not assume that PWSs are already in compliance with state standards.	This approach overestimates costs for water systems in states with existing state standards. The EPA adjusts the baseline by setting the maximum pre-regulation concentrations equal to the state MCL for systems in states with promulgated regulations. This allows the EPA to capture the incremental costs of the NPDWR MCLs more accurately.
Nontreatment options	Assumes all exceeding EP will install a treatment technology to comply with the MCLs.	This assumption overestimates costs, as the EPA is aware of a number of water systems that have elected to drill a new well to reduce PFAS concentrations in supplied water. Another commenter pointed out that Michigan expects up to 26 percent of water systems to interconnect with other systems to comply with their state standard. Other commenters pointed out the viability of interconnection and new wells as compliance options will vary regionally, and the EPA agrees. Nevertheless, the absence of these options entirely in AWWA's B&V report overestimates national costs.

Table 25. EPA Response to key capital cost assumptions in AWWA B&V Report

Analytical Component	**AWWA B&V report**	**EPA response**
Equipment lifespan	Assumes a fixed life cycle cost using a fixed 20-year lifespan for all capital equipment.	A 20-year lifespan may be reasonable for very small systems but based on the composite useful life of treatment systems derived from the useful lives of individual treatment system components and industry information, the EPA estimates that treatment system useful life can be 30 years or more for medium to larger systems using more durable materials of construction.
Contingency factors	Includes a contingency factor of 4 percent under contractor markup and an additional contingency factor of 30 percent under non-construction costs.	The inclusion of contingency twice is unusual and may not reflect actual realized contingency costs at project completion. A Construction Industry Institute (2001) study found that projects of $100 million or less incurred only 74 percent or less of the contingency initially budgeted. The EPA updated its approach to incorporate a contingency factor of 5 to 10 percent depending on total project cost at all cost levels for systems installing treatment. The EPA also included a miscellaneous allowance of 10 percent, which can be considered a form of contingency.
Building costs	Assumes a fixed unit cost of $200/square foot for buildings.	AWWA's fixed unit cost likely overestimates actual building costs, particularly for small systems that may not require complex or architecturally detailed buildings. The EPA estimates that building costs vary depending on building quality and square footage and range from $57/square foot to $204/square foot.
Pumping and backwash assumptions	Assumes that all GAC and IX treatment systems require a new influent pumping station, and all GAC and IX treatment systems require new backwash pumps. Except for the two smallest size categories, assumes all GAC and IX treatment systems require backwash recovery basins providing 20 feet of water depth.	AWWA's assumptions overestimate costs as many systems, including small groundwater systems, likely have sufficient existing influent pumping pressure to cover the additional head loss. Some systems using GAC (especially small systems) may not need a dedicated new backwash pump and may be able to accomplish backwash using existing influent or treated water pumps. In applications using PFAS-selective IX resins, periodic backwashing is not recommended (Berretta et al., 2021), so the need for these pumps is questionable and the assumption overestimates costs.

Capital equipment costs	The Association of Metropolitan Water Agencies (AMWA) and the AWWA surveyed its members to obtain recent cost data on installed PFAS treatment systems at drinking water treatment plants.	The EPA updated its equipment costs to 2022 dollars using current price indices. The EPA also collected new vendor price quotes for cost driver equipment components (e.g., pressure vessels, treatment media) and made several other adjustments to WBS model assumptions about pilot study costs and contingency costs that increased total capital costs. The B&V model, as presented in Figure 7-1 of AWWA's public comment letter, appears to overestimate costs for many of the case studies included in the B&V report. For example, it results in higher costs for 28 of the 32 case studies (88 percent) shown in Figure 7-1. The EPA assessed the WBS model results in comparison to the costs of GAC equipment packages from 2023 supplied by a nationally recognized vendor of GAC media and GAC treatment systems. Based on this assessment, the EPA concluded that the direct capital costs in the WBS model for comparable packages of equipment, excluding items the vendor does not supply, range from 23 percent lower to 19 percent higher than the vendor costs and with two exceptions, they are within 10 percent of the vendor costs.
Small system capital costs	Listed capital costs for small systems ranging from $900,000 to $5,300,000.	The EPA accounts for the use of package systems. AWWA appendix B, Table 3-1, indicates that their pressure GAC model accepts treatment capacity inputs from 1 to 12 mgd. It does not indicate how the model handles design flows less than 1 mgd. It is possible that the parametric estimates the model uses are not a good fit below this threshold and does not account for the use of package systems.
Average and design flow estimates	Service population data from SDWIS was used and the average flow for each PWS was assumed based on a per capita per day usage of 150 gallons. Peaking factors for different size systems from the EPA's *Cost and Technology Document for Final Groundwater Rule* were used.	Estimated design flow of a water system effects the size and cost of the capital equipment that will be installed on site. Average flow estimates are the driver for many operational costs. AWWA's approach to estimating design and average flow requirements overestimates the treatment system flow requirements, particularly for smaller systems. For the smallest systems, AWWA's approach overestimates flows by up to 30 percent. The EPA estimated the average daily flow and design flow for drinking water systems based on the empirical relationship between retail population served and flow. This relationship was derived using the data collected via the

		CWSS. It is reported in the EPA's *Geometries and Characteristics of Public Water Systems* report (USEPA, 2000g). As detailed in Table 4-34 of the EA for the final rule, water use efficiency has increased substantially since these relationships were developed, and therefore the trend of lower residential water use could result in lower flow per population and lower treatment costs as compared to predicted values in the EPA's analysis.

Table 26. EPA Response to key operation and maintenance cost assumptions in AWWA B&V Report

Analytical Component	AWWA B&V report	EPA response
Bed life	The BV values utilized for GAC were derived from data collected during a Black & Veatch GAC pilot study for Cape Fear Public Utility Authority (CFPUA). The values utilized for IX were derived partially from data collected during a Black & Veatch IX pilot study for CFPUA and partially from data collected during an IX pilot study for La Habra Height County Water District.	AWWA estimates bed life for all systems using parameters derived from one or two pilot studies. These site-specific pilot studies may not be representative of the range of water quality conditions experienced by systems across the country. For GAC in particular, using the parameters in AWWA's Table 5-9 results in estimated bed lives of less than 7,000 and 9,000 BVs for 90 percent removal of PFOA and PFOS, respectively. These short bed life estimates result in high annual operating costs and may be an artifact of the relatively high influent TOC in the CFPUA pilot study that is the basis of AWWA's estimates. Surface and groundwater systems with more moderate to low influent TOC would be expected to experience much longer GAC bed life and lower operating costs.
Disposal of treatment media	Assumed that spent GAC media would be incinerated "because of the unknown viability of GAC media reactivation under CERCLA." Replacement costs were therefore assumed to be virgin media.	The EPA has proposed PFOA and PFOS be designated as hazardous substances under CERCLA. If finalized, the designation of PFOA and PFOS as CERCLA hazardous substances would not require waste (e.g., biosolids, treatment residuals, etc.) to be treated in any particular fashion, nor disposed of at any specific particular type of landfill. The designation also would not restrict, change, or recommend any specific activity or type of waste at landfills. This action should not result in limiting disposal options and how PFAS containing waste, including spent GAC or resin, is required to be managed. However, drinking water treatment operations may choose to send spent GAC and resin containing PFAS to facilities permitted to treat and/or dispose of hazardous wastes. Even where reactivation is not feasible, disposal in a RCRA permitted hazardous waste disposal facility is expected to be a more cost-effective option than incineration. Therefore, the assumption of incineration and replacement with virgin media overestimates the disposal costs in the B&V report.

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c. Treatment Costs in the Final Rule Analysis

The cost estimates in the EA for the final PFAS NPDWR reflects the adjustments made to the WBS curves and decision tree based on public comments discussed above as well as the additional occurrence information available since the publication of the proposed PFAS NPDWR. For detailed information on the EPA's occurrence analysis, see section VI of this preamble. For detailed information on the EPA's cost analysis and the EPA's estimates of the national annualized costs of the final MCLs, see section XII.D.

3. Primacy Agency Costs

a. Primacy Agency Cost Estimates in the Proposal

In the EA for the proposed rule, the EPA estimated the costs incurred by primacy agencies associated with the rule, including up front implementation costs as well as costs associated with system actions related to sampling and treatment.

b. Summary of Major Public Comments on Primacy Agency Costs and EPA Responses

Many commenters state that the EPA has underestimated the costs to primacy agencies required to comply with the rule. One commenter stated, ''EPA's analysis of primacy agency costs does not accurately capture all the activities that primacy agencies will undergo for PFAS implementation and underestimates the number of hours for the primacy tasks.'' Commenters recommend that the EPA use findings from ASDWA's PFAS Cost of State Transactions Study (PCoSTS) to reevaluate the primacy agency costs estimated in the EA. The EPA's response to specific recommendations is discussed here.

The EPA agrees with commenters on the burdens associated with regulatory start up; primacy package adoption; technical, managerial, and financial (TMF) assistance to water systems; and reviewing and approving treatment. Commenters pointed out activities not explicitly accounted for in the regulatory start up estimate in the EA for proposal including accreditation of laboratories for PFAS testing; SDWIS updates; monitoring schedule updates; time spent responding to questions from members of the public; inquiries from public officials; and media requests immediately following the final publication of the NPDWR. Commenters also pointed out that adopting primacy packages is a significant undertaking with ''specific and very detailed administrative procedures that must be adhered to in order to adopt water quality regulations'' and that ''some primacy agencies have requirements for robust public comment periods as a component of new rule adoption.'' As recommended by commenters, the EPA created a new cost item for primacy package adoption. Commenters stated the EPA's assumption in the proposal that the amount of time a primacy agency will need to review treatment plans directly correlates with the size of the water system was inaccurate. Commenters noted that ''. . . small systems often take the most time as they need significant assistance to navigate the process for the design and construction of new treatment and get into compliance.'' After considering these comments, the EPA agrees that reviewing and approving treatment for small systems is likely to take more time given the assistance needed for these systems. Because small systems often lack the technical, managerial, and financial capacity, it is likely that primacy agencies will spend more time assisting these systems in navigating compliance with the PFAS NPDWR. As such, the EPA adjusted burden estimates in the final rule to reflect the largest primacy agency burden per EP at the smallest systems and decreased burden hours with increasing system size, as commenters suggested.

Several commenters disagreed with the EPA's exclusion of additional costs to primacy agencies associated with reporting regarding violations, variances and exemptions, enforcement actions, and other compliance related primacy agency activities in the national cost analysis. One commenter estimated the PFAS NPDWR will likely result in hundreds of violations once in effect. The EPA recognizes that these activities do have an associated burden for primacy agencies but disagrees that these costs should be included in the EA. The EPA assumed 100 percent compliance for its national level analysis in the EA for the final rule because the EPA has determined that the final rule is feasible given known occurrence concentrations and efficacy of the technologies available. Further, this is consistent with the approach taken in EAs for other NPDWRs (USEPA, 2005c; USEPA, 2019c; USEPA, 2020f). Commenters recommended that the EPA include hours for additional annual reporting. The EPA disagrees and expects that adding PFAS results to already-required reports will have no discernable incremental burden for quarterly or annual reports to SDWIS Fed.

Commenters recommended that the EPA include the costs associated with various compliance activities. Given the EPA's assumption of 100 percent compliance for its national level analysis in the EA discussed above, the EPA disagrees and did not take commenters' recommendations to include the costs associated with assisting out of compliance systems and assisting systems to remain in compliance, pursuing enforcement actions, staff time checking in with system violations and reviewing system variances and exemptions. The EPA did include the costs associated with compliance activities for systems in compliance, including updating inspection SOPs and additional sanitary survey burden at water systems that have installed treatment to comply with the PFAS NPDWR.

c. Primacy Agency Costs in the Final Rule Analysis

After considering public comments on the burden hours associated with primacy agency activities, the EPA made the following changes. The EPA increased the estimate from 416 hours to ''read and understand the rule as well as adopt reg requirements'' to 4,000 hours per primacy agency to conduct a suite of regulatory start up activities. Per commenters' recommendation, the EPA included a new line item for primacy package adoption and estimated 300 hours per primacy agency. The EPA lowered the water system operator TMF training from 2,080 hours to 1,500 hours per primacy agency based on commenter recommendations. The EPA added a one-time burden estimate of 20 hours to inspection SOPs and an additional 2–5 burden hours for the primacy agency, by water system size, per sanitary survey per system installing treatment to comply with the rule. For more information see section XII.D.

4. Costs of the Hazard Index

a. Hazard Index Cost Estimates in the Proposal

In the EA for the proposed rule, the EPA estimated national costs associated with PFOA, PFOS, and PFHxS. Given available occurrence data for the other compounds in the proposed rule (PFNA, HFPO–DA, and PFBS) and the regulatory thresholds under consideration, the EPA did not use SafeWater to model national costs associated with potential Hazard Index (HI) exceedances as a direct result of these contaminants. To assess the potential impact of these compounds in the proposed rule, the EPA conducted an analysis of the additional, or incremental, system level impact that occurrence of these contaminants would have on treatment costs. The EPA estimated that the Hazard Index would increase costs by 0–77 percent at the system level, with costs varying due to PFAS occurrence scenario and treatment technology used.

b. Summary of Major Public Comments on Hazard Index Costs and EPA Responses

A few commenters recommended that the EPA further consider the costs associated with compliance with the Hazard Index (HI) MCL. Specifically, commenters stated that the EPA's analysis of system level costs associated with the Hazard Index does not adequately characterize the overall costs that will be incurred due to the Hazard Index standard. One commenter stated that ''EPA should not move forward with the Hazard Index until it has satisfied its statutory and policy obligation to conduct a cost-benefit analysis.'' Some commenters voiced concern regarding the EPA's assumption that costs associated with compliance with the Hazard Index MCL are insignificant and asserted that these costs must be reexamined, stating that this assessment ''requires more knowledge on the nationwide occurrence of these compounds'' and that the EPA ''cannot assume that addressing the costs of PFOA and PFOS is sufficient when the additional four PFAS will be driving treatment decisions at some PWSs.'' Conversely, one commenter asserted that available occurrence data demonstrate that few systems will be required to install treatment to comply with the Hazard Index MCL that would not already be treating to comply with the PFOA and PFOS MCLs.

The EPA disagrees with commenters who state that the agency did not meet its requirements under SDWA, which requires the agency to analyze ''quantifiable and nonquantifiable costs . . . that are likely to occur solely as a result of compliance with the maximum contaminant level.'' In the proposal, the EPA analyzed the quantifiable costs of the Hazard Index at the system level, using the best available information at the time of publication, and analyzed the nonquantifiable costs of the Hazard Index by including a qualitative discussion of the national level impacts and therefore met the statutory requirements under SDWA 1412(b)(3)(C). After considering recommendations from the public comments to further analyze the costs of the Hazard Index and the data available to support a quantitative analysis of the costs of the Hazard Index, the EPA decided to conduct a sensitivity analysis of the costs of the Hazard Index at the national level. The results of the sensitivity analysis supported the EPA's assumption in the proposal that quantified national costs are marginally underestimated as a result of this lack of sufficient nationally representative occurrence data. The EPA's consideration of Hazard Index costs in the final rule analysis are discussed in the following subsection.

c. Hazard Index and PFHxS, PFNA, and HFPO–DA MCL Costs in the Final Rule Analysis

To estimate quantified costs of the final rule presented in the national-level summary tables, the EPA first estimated baseline PFAS occurrence using a Bayesian hierarchical model fitted with sampling data collected from systems participating in UCMR 3. The model included three of the six PFAS compounds regulated through this NPDWR: PFOA, PFOS, and PFHxS (see section VI of this preamble). This permitted the agency to quantify costs at a national level with a higher degree of confidence and precision for these three PFAS than if simple extrapolations had been used. Since there are some limitations with nationally representative occurrence information for the other compounds that were either not included in UCMR 3 (HFPO–DA) or did not have a sufficient number of observed values above the UCMR 3 reporting limits (PFNA, PFBS), the EPA has a lesser degree of confidence and precision for its quantified estimates of these three PFAS, which are informed by a significant amount of available state-level data. Therefore, the EPA presented the cost estimates for PFNA, HFPO–DA, and PFBS in a sensitivity analysis in the EA (*i.e.,* national-level sensitivity analysis, see appendix N.3) instead of including these costs in the summary tables of quantified national level costs.[16]

In the EA for the proposed PFAS NPDWR, the EPA used a model system approach [17] to illustrate the potential incremental costs for removing PFAS not included in the national economic model (*i.e.,* PFNA, HFPO–DA, and PFBS). After considering public comments on the incremental cost analysis, many of which encouraged the EPA to further evaluate and consider quantified costs of the Hazard Index MCL where feasible, the EPA updated and combined existing analyses contained in the rule proposal to evaluate the incremental costs associated with the Hazard Index MCL and individual MCLs for PFNA and HFPO–DA with a quantified national level sensitivity analysis in the final rule. The updated analysis for the final rule builds on the proposal analysis by combining information that was presented separately at proposal. The analysis in appendix N of the final EA utilizes the system level treatment cost information presented at proposal (See appendix N of USEPA, 2023n, 2023o) with updates to the cost models for the final rule detailed in section XII.A.2. These treatment costs were applied to the number of systems expected to exceed the standards based on PFNA, PFBS, and HFPFO–DA occurrence using the approaches for estimating occurrence of these compounds presented at proposal (see section 10.3 of USEPA, 2023l). This modified analysis was primarily conducted to ensure that the EPA has not, as some commenters claim, substantially underestimated the potential magnitude of these costs. The EPA notes the approach presented in appendix N for the final rule and summarized here, by connecting analyses for proposed rule, allows the agency to consider and compare the relative degree of the potential overall costs of these otherwise nonquantifiable costs of the Hazard Index and PFNA and HFPO–DA MCLs relative to overall national rule costs. This analysis confirms the EPA's findings at proposal that the Hazard Index costs (and those costs for regulating PFNA and HFPO–DA individually) make up a small portion of

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[16] When available, nationally representative occurrence information is preferable for an economic analysis of national level costs and benefits. In the case of PFOA, PFOS, and PFHxS, the EPA has a sufficiently robust nationally representative dataset from UCMR 3. The EPA used additional state data that were available at systems that were part of this UCMR 3 set of systems to fit the national occurrence model that informed cost estimates for PFOA, PFOS, and PFHxS (see Cadwallader et al., 2022). In the case of PFNA, HFPO–DA, and PFBS, the EPA lacks the same level of precision as described above for PFOA, PFOS, and PFHxS. State-led data collection efforts provided valuable information about occurrence for PFNA, HFPO–DA, and PFBS, however they did not provide the nationally representative foundation provided by UCMR3 for PFOA, PFOS, and PFHxS to be incorporated into the MCMC national occurrence model.

[17] At proposal, the EPA used a model system approach for estimating potential incremental treatment costs associated with co-occuring PFAS at systems already required to treat in the national model framework and the potential per system costs for the set of systems triggered into treatment as a result of Hazard Index MCL exceedances not already captured in the national analysis. For further detail on the assumptions and findings of the EPA's analysis of incremental costs of other PFAS at rule proposal, please see appendix N.3 in the Economic Analysis for the Proposed Per- and Polyfluoroalkyl Substances National Primary Drinking Water Regulation (USEPA, 2023n, 2023o).

the overall rule costs. Likewise, the EPA notes that while these costs are presented in appendix N because of the lesser degree of confidence and precision in the estimates, the EPA has considered these costs as part of this final regulation. It has done so by evaluating nonquantifiable costs and accounting for uncertainty, characterizing these otherwise nonquantifiable costs in appendix N to generate cost estimates that, while useful, are not as statistically robust as the national cost estimates presented in chapter 5 of the EA. Using this analysis, the agency has confirmed the Hazard Index and PFNA and HFPO–DA MCLs drive a relatively low percentage of the overall rule costs. The EPA has also considered these costs in the context that the Hazard Index and PFHxS, PFNA, and HFPO–DA MCLs are expected to deliver important nonquantifiable health benefits, including PFNA birth weight benefits[18] and other nonquantifiable benefits associated with the reduction of the Hazard Index PFAS (PFNA, PFHxS, HFPO–DA, and PFBS)[19] described in chapter 6.2 of the EA.

The proposed rule included a Hazard Index MCLG and MCL for any mixture of one or more of PFHxS, HFPO–DA, PFNA, and PFBS. The final rule includes a Hazard Index MCLG and MCL for any mixture of two or more of PFHxS, HFPO–DA, PFNA, and PFBS. The final rule also includes individual MCLGs and MCLs for PFHxS, PFNA, and HFPO–DA. The EPA's cost analysis at proposal considered the costs associated with the individual MCLs for PFHxS, PFNA, and HFPO–DA because the proposed Hazard Index MCL would function as individual MCLs when these contaminants occur in isolation. While the rule structure has changed in the final NPDWR, the costing framework used at proposal is still applicable in the final rule: what was considered a Hazard Index MCL exceedance at proposal would be an individual MCL exceedance under the final rule should those contaminants occur in isolation. Further, a Hazard Index exceedance in the final rule (defined as two or more of PFHxS, PFNA, HFPO–DA, and PFBS) is unchanged from a costing perspective to what the EPA proposed. Whether a system exceeds a Hazard Index MCL or individual MCL in the final rule, these costs are captured in the cost estimates the EPA considered and presented in appendix N.3 of the EA and summarized in this section. Specifically, if a system exceeds only one of the individual MCLs for PFHxS, PFNA, or HFPO–DA that exceedance is costed by estimating the removal needed to achieve compliance with a given individual MCL. If a system exceeds the Hazard Index MCL, that exceedance is costed by estimating the removal of the combination of contaminants needed to achieve compliance with the Hazard Index MCL. Therefore, the national level cost estimate for PFHxS is reflective of both the total national cost of the PFHxS individual MCL and instances of Hazard Index MCL exceedances where PFHxS is present above its HBWC while other Hazard Index PFAS are present.

To understand the totality of national-level cost impacts for the Hazard Index MCL, the EPA considered both the contribution of PFHxS (estimated as part of the national level cost analysis), as well as the costs for PFNA, HFPO–DA, and PFBS (estimated in the appendix N sensitivity analysis). Together, these provide information on the costs for the Hazard Index MCL and the individual MCLs for PFHxS, PFNA, and HFPO–DA, as a whole. Due to available data informing the Bayesian hierarchical occurrence model, the EPA was only able to quantify the portion of total costs for the Hazard Index MCL attributable to PFHxS[20] in the national level analysis. The EPA notes that this estimate also represents the national level quantified costs for the individual PFHxS MCL. The EPA acknowledges that this $11.6 million estimate is only a portion of the costs imposed by the Hazard Index MCL and also does not account for the costs imposed by the individual PFNA and HFPO–DA MCLs. The EPA accounted for those potential additional costs through the sensitivity analysis described in appendix N, in which the EPA found that costs of treating for PFNA, HFPO–DA, and PFBS to meet the Hazard Index MCL and individual MCLs for PFNA and HFPO–DA increased national costs by approximately 5 percent, from $1,549 million to $1,631 million. These costs represent the total costs of the final rule; in other words, this includes the costs associated with individual MCLs for PFOA, PFOS, PFHxS, HFPO–DA, and PFNA, as well as the Hazard Index MCL. Due to data limitations, the EPA has not separately estimated the costs of the Hazard Index in the absence of the individual MCLs. The sensitivity analysis demonstrates that the quantified national analysis cost estimate that includes only PFOA, PFOS, and PFHxS (where PFHxS represents only a portion of the Hazard Index costs) marginally underestimates total rule costs when also considering the potential cost impacts attributable to HFPO–DA, PFNA, and PFBS. The cost estimates stemming from both the quantified national estimate for PFOA, PFOS, and PFHxS, and from the sensitivity analysis conducted for PFNA, HFPO–DA, and PFBS together inform the impact of the Hazard Index MCL as required by the HRRCA under SDWA.

To fully weigh the costs and benefits of the action, the agency considered the totality of the monetized values, the potential impacts of the nonquantifiable uncertainties, the nonquantifiable costs and benefits, and public comments received by the agency related to the quantified and qualitative assessment of the costs and benefits. For the final rule, the EPA is reaffirming the Administrator's determination made at proposal that the quantified and nonquantifiable benefits of the rule justify its quantified and nonquantifiable costs.

In light of the individual MCLs, the EPA has separately presented national level marginal costs associated with the individual MCLs for PFHxS, PFNA and HFPO–DA in the absence of the Hazard Index MCL; see chapter 5.1.3 and appendix N.4 of the EA for details. Therefore, the costs for the individual PFHxS, PFNA, and HFPO–DA MCLs have been considered both in the

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[18] As discussed in appendix K.4, a 1 ppt reduction in both PFOA and PFOS for a system serving a population of 100,000 would result in $0.101 million in annualized birth weight benefits. If including a 1 ppt PFNA reduction, in addition to a 1 ppt reduction in both PFOA and PFOS, for a system serving a population of 100,000, the resulting annualized birth weight benefits would increase by $0.464 to $0.689 million, depending on the slope factor used for PFNA. The EPA estimates that 208 water systems may exceed the PFNA MCL.

[19] The EPA also anticipates additional substantial benefits to PWS customers associated with reduced exposure to Hazard Index compounds (PFHxS, HFPO–DA, PFNA, and PFBS) not included in the primary analysis. The nonquantifiable benefits impact categories include developmental, cardiovascular, immune, hepatic, endocrine, metabolic, reproductive, musculoskeletal, and carcinogenic effects. See chapter 6.2 of the EA for more information.

[20] The EPA notes that there are anticipated to be circumstances where PFHxS exceeds its individual MCL and HBWC where PFNA, PFBS, and HFPO–DA do not co-occur. While resulting in an exceedance of the PFHxS MCL, if PFHxS exceeds its HBWC without other Hazard Index PFAS present, this would not result in an exceedance of the Hazard Index MCL. At rule proposal, a single exceedance of any of the four Hazard Index PFAS would have resulted in an exceedance of the Hazard Index MCL. However, to improve rule implementation and to support effective risk communication, the EPA has structured the final rule such that a Hazard Index exceedance only occurs when there are two or more of the Hazard Index PFAS present. Therefore, while for purposes of informing its quantified cost analysis the EPA is assuming that every PFHxS exceedance of the MCL also causes an exceedance of the Hazard Index MCL, this approach results in the EPA overestimating PFHxS-attributable Hazard Index costs in its national cost analysis.

proposed and final rule. For more information on the agency's methodology, findings, and limitations of the EPA's updated analysis of costs associated with compliance with the Hazard Index, please see appendix N.3 of the EA (USEPA, 2024e).

5. Benefit-Cost Determination

a. Benefit-Cost Determination in the Proposal

When proposing an NPDWR, the Administrator shall publish a determination as to whether the benefits of the MCL justify, or do not justify, the costs based on the analysis conducted under section 1412(b)(3)(C). For the proposed rule, the Administrator determined that the quantified and nonquantifiable benefits of the proposed PFAS NPDWR justified the costs.

b. Summary of Major Public Comments on Benefit-Cost Determination and EPA Responses

Many commenters agreed with the Administrator's determination that the benefits of the rule justify its costs. Specifically, commenters asserted that the EPA's estimation of the net benefits of enacting the MCLs is reasonable, stating that ''even if the costs are very substantial, the benefits associated with the anticipated drinking water improvements justify such expenditures.'' Commenters also stated that it is likely that ''the analysis understates the benefits'' of the rule, particularly given the ''significant unquantified risk reduction benefits and co-benefits'' that are anticipated to result from the rule.

In response to these comments, the EPA agrees that its quantified benefits likely significantly understate the benefits of the rule due to the large share of nonquantifiable benefits that are expected to be realized as avoided adverse health effects, in addition to the benefits that the EPA has quantified. The EPA anticipates additional benefits associated with developmental, cardiovascular, liver, immune, endocrine, metabolic, reproductive, musculoskeletal, and carcinogenic effects beyond those benefits associated with decreased PFOA and PFOS that the EPA has quantified. In response to commenters urging the EPA to quantify additional health endpoints associated with PFAS exposure, the EPA has developed a quantitative sensitivity analysis of PFOS effects and liver cancer, further strengthening the justification for this determination. Due to occurrence, health effects, and/or economic data limitations, the EPA is unable to quantitatively assess additional benefits of the rule.

Conversely, several commenters stated that the EPA has failed to demonstrate that the benefits of the rule justify its costs. Specifically, commenters disagreed with this determination because the EPA's analysis ''significantly underestimates the costs of the proposed MCLs. . .and overestimates its benefits.'' Commenters asserted that the EPA needs to update its EA to more accurately reflect the true costs of compliance of the rule to make the determination that the rule's costs are justified by its benefits. A few commenters urged the EPA to consider whether the benefits of finalizing the rule at regulatory alternative MCLs (*e.g.,* 5.0 or 10.0 ng/L) would better justify the costs of the rule.

After considering public comments, the EPA has made a number of adjustments to the cost model and collectively these changes have increased the agency's estimated annualized costs. The EPA has used the best available peer reviewed science to inform the cost estimates, including treatment costs, of the final PFAS NPDWR. For more information on the EPA's responses to comments on the rule costs, see sections XII.A.2–XII.A.4 of this preamble. The EPA disagrees with commenters that the EPA has overstated the benefits. As discussed in section XII.A.1, the EPA has used the best available peer reviewed science to quantify the benefits of the rule. The EPA also disagrees with commenters that suggested the benefits ''better justify'' the costs of PFOA and PFOS standards at 5.0 or 10.0 ng/L. These commenters pointed to the quantified net benefits of the regulatory alternatives and noted that net benefits are positive at 3 and 7 percent discount rates for a standard of 10.0 ng/L for PFOA and PFOS. The commenters' sole reliance on the quantified costs and benefits of the rule to support their argument is incorrect, as SDWA requires the agency to consider both the quantifiable and nonquantifiable impacts of the rule in the determination. Under SDWA 1412(b)(4)(B), the EPA is required to set an MCL as close as feasible to the MCLG, taking costs into consideration. In other words, SDWA does not mandate that the EPA establish MCLs at levels where the quantified benefits exceed the quantified costs. This was many commenters' justification for the recommendation to promulgate a standard of 10.0 ppt each for PFOA and PFOS in lieu of the proposed rule, and the EPA therefore disagrees that quantified costs and benefits can or should be the sole determinant of an MCL value. The Administrator's assessment that the benefits of the proposed rule justified its costs was based on the totality of the evidence, specifically the quantified and nonquantifiable benefits, which are anticipated to be substantial, as well as the quantified and nonquantifiable costs. Other commenters incorrectly stated that SDWA requires the EPA to set an MCL at a level '' . . . that maximizes health risk reduction benefits at a cost that is justified by the benefits.'' This test is found in section 1412(b)(6)(A) of SDWA and applies only when the Administrator determines based on the HRRCA that the benefits of a proposed MCL developed in accordance with paragraph (4) would not justify the costs of complying with the level. In the case of the proposed PFAS NPDWR, the Administrator determined that the benefits justify the costs for MCLs set as close as feasible to the MCLGs. For more information on the EPA's response to comments on the regulatory alternative MCLs considered in this rule, see section V of this preamble.

c. Benefit-Cost Determination in the Final Rule Analysis

For the final rule, considering both quantifiable and nonquantifiable costs and benefits of the rule as discussed in the EA and EA Appendices, the EPA is reaffirming the Administrator's determination made at proposal that the quantified and nonquantifiable benefits of the MCLs justify their costs.

B. Affected Entities and Major Data Sources Used To Develop the Baseline Water System Characterization

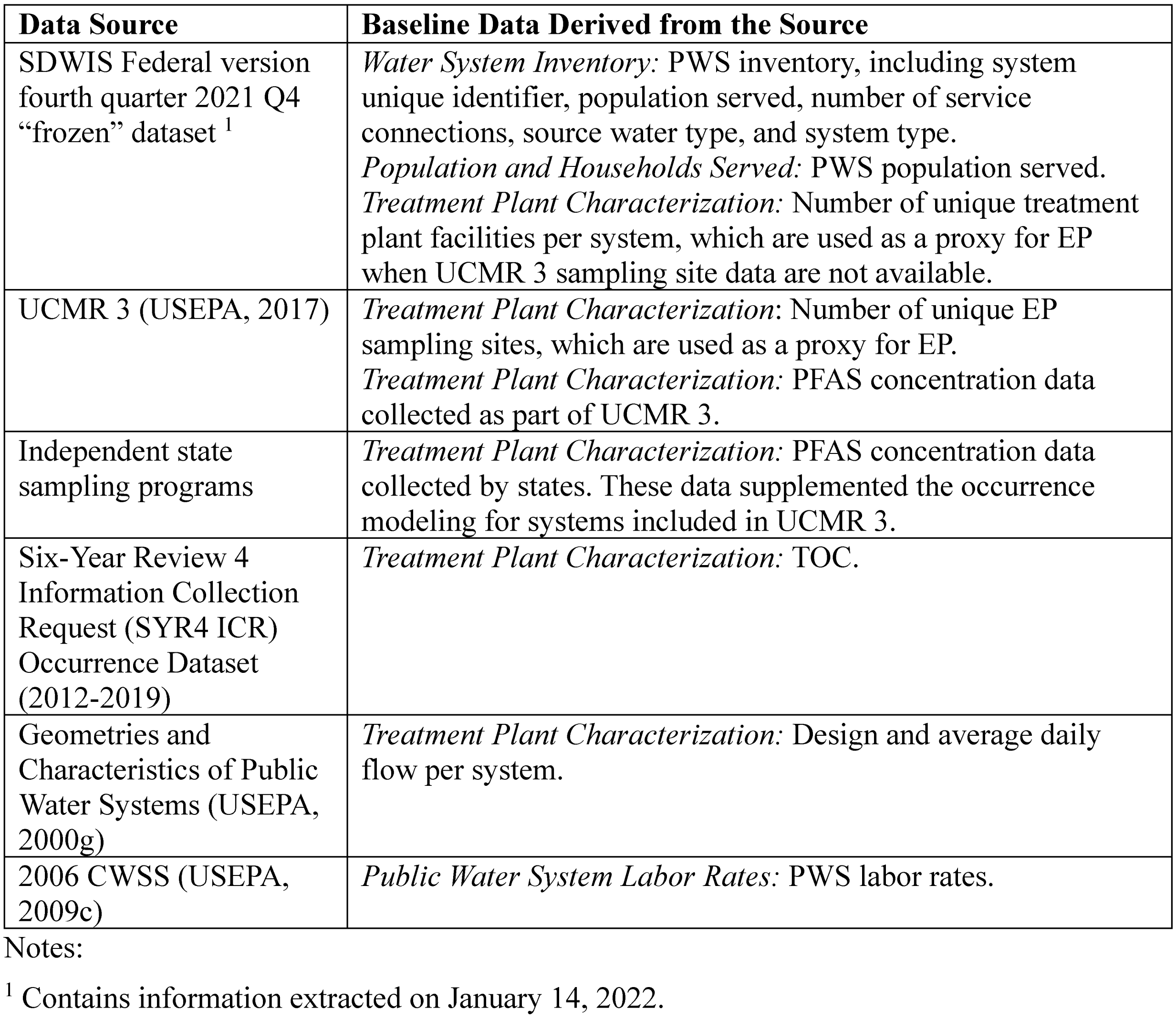
The entities potentially affected by the final rule are primacy agencies and PWSs. PWSs subject to final rule requirements are either CWSs or NTNCWSs. These water systems can be publicly or privately owned. PWSs subject to the rule would be required to meet the MCL and comply with monitoring and reporting requirements. Primacy agencies would be required to adopt and enforce the drinking water standard as well as the monitoring and reporting requirements.

Both PWSs and primacy agencies are expected to incur costs, including administrative costs, monitoring, and reporting costs, and in some cases, anticipated costs to reduce PFAS levels in drinking water to meet the final rule using treatment or nontreatment options. Section D of this preamble summarizes the method the EPA used to estimate these costs.

The systems that reduce PFAS concentrations will reduce associated health risks. The EPA developed methods to estimate the potential benefits of reduced PFAS exposure among the service populations of systems with PFAS levels exceeding the final drinking water standard. Section E summarizes the method used to estimate these benefits.

In its *Guidelines for Preparing Economic Analyses,* the EPA characterizes the ''baseline'' as a reference point that reflects the world without the final regulation (USEPA, 2016e). It is the starting point for estimating the potential benefits and costs of the final NPDWR. The EPA used a variety of data sources to develop the baseline drinking water system characterization for the regulatory analysis. Table 27 lists the major data sources and the baseline data derived from them. Additional detailed descriptions of these data sources and how they were used in the characterization of baseline conditions can be found in chapter 4 of USEPA (2024g).

Table 27: Data Sources Used to Develop Baseline Water System Characterization



Data Source	Baseline Data Derived from the Source
SDWIS Federal version fourth quarter 2021 Q4 "frozen" dataset [1]	*Water System Inventory:* PWS inventory, including system unique identifier, population served, number of service connections, source water type, and system type. *Population and Households Served:* PWS population served. *Treatment Plant Characterization:* Number of unique treatment plant facilities per system, which are used as a proxy for EP when UCMR 3 sampling site data are not available.
UCMR 3 (USEPA, 2017)	*Treatment Plant Characterization*: Number of unique EP sampling sites, which are used as a proxy for EP. *Treatment Plant Characterization:* PFAS concentration data collected as part of UCMR 3.
Independent state sampling programs	*Treatment Plant Characterization:* PFAS concentration data collected by states. These data supplemented the occurrence modeling for systems included in UCMR 3.
Six-Year Review 4 Information Collection Request (SYR4 ICR) Occurrence Dataset (2012-2019)	*Treatment Plant Characterization:* TOC.
Geometries and Characteristics of Public Water Systems (USEPA, 2000g)	*Treatment Plant Characterization:* Design and average daily flow per system.
2006 CWSS (USEPA, 2009c)	*Public Water System Labor Rates:* PWS labor rates.

Notes:

[1] Contains information extracted on January 14, 2022.

C. Overview of the Cost-Benefit Model

The EPA's existing SafeWater Cost Benefit Model (CBX) was designed to calculate the costs and benefits associated with setting a new or revised MCL. Since the final rule simultaneously regulates multiple PFAS contaminants, the EPA developed a new model version called the SafeWater Multi-Contaminant Benefit Cost Model (MCBC) to efficiently handle more than one contaminant. SafeWater MCBC allows for inputs that include differing mixtures of contaminants based on available occurrence data as well as multiple regulatory thresholds. The model structure allows for assignment of compliance technology or technologies that achieve all regulatory requirements and estimates costs and benefits associated with multiple PFAS contaminant reductions. SafeWater MCBC is designed to model co-occurrence, sampling, treatment, and administrative costs, and simultaneous contaminant reductions and resultant benefits. The modifications to the SafeWater model are consistent with the methodology that was developed in the single MCL SafeWater CBX Beta version that was peer reviewed. More detail on the modifications to the SafeWater model can be found in section 5.2 of the EPA's EA.

The costs incurred by a PWS depend on water system characteristics; SDWIS

Fed provides information on PWS characteristics that typically define PWS categories, or strata, for which the EPA developed cost estimates in rulemakings, including system type (CWS, NTNCWS), number of people served by the PWS, the PWS's primary raw water source (ground water or surface water), the PWS's ownership type (public or private), and the state in which the PWS is located.

Because the EPA does not have complete PWS-specific data across the approximately 49,000 CWSs and 17,000 NTNCWSs in SDWIS Fed for many of the baseline and compliance characteristics necessary to estimate costs and benefits, such as design and average daily flow rates, water quality characteristics, treatment in-place, and labor rates, the EPA adopted a ''model PWS'' approach. SafeWater MCBC creates model PWSs by combining the PWS-specific data available in SDWIS Fed with data on baseline and compliance characteristics available at the PWS category level. In some cases, the categorical data are simple point estimates. In this case, every model PWS in a category is assigned the same value. In other cases, where more robust data representing system variability are available, the category-level data include a distribution of potential values. In the case of distributional information, SafeWater MCBC assigns each model PWS a value sampled from the distribution. These distributions are assumed to be independent.

For a list of PWS characteristics that impact model PWS compliance costs, please see chapter 5 of USEPA (2024g). These data include inventory data specific to each system and categorical data for which randomly assigned values are based on distributions that vary by category (*e.g.,* ground water and surface water TOC distributions or compliance forecast distributions that vary by system size category).

Once model PWSs are created and assigned baseline and compliance characteristics, SafeWater MCBC estimates the quantified costs and benefits of compliance for each model PWS under the final rule. Because of this model PWS approach, SafeWater MCBC does not output any results at the PWS level. Instead, the outputs are cost and benefit estimates for 36 PWS categories, or strata. Each PWS category is defined by system type (CWS and NTNCWS), primary water source (ground or surface), and size category. Note the EPA does not report state-specific strata although state location is utilized in the SafeWater MCBC model (*e.g.,* current state-level regulatory limits on PFAS in drinking water). The detailed output across these strata can be found in the chapter 5 of USEPA (2024g).

For each PWS category, the model then calculates summary statistics that describe the costs and benefits associated with final rule compliance. These summary statistics include total quantified costs of the final rule, total quantified benefits of the final rule, the variability in PWS-level costs (*e.g.,* 5th and 95th percentile system costs), and the variability in household-level costs.

D. Method for Estimating Costs

This section summarizes the cost elements and estimates total cost of compliance for the PFAS NPDWR discounted at 2 percent. The EPA estimated the costs associated with monitoring, administrative requirements, and both treatment and nontreatment compliance actions associated with the final rule (USEPA, 2024g).

1. Public Water System (PWS) Costs

a. PWS Treatment and Nontreatment Compliance Costs

The EPA estimated costs associated with engineering, installing, operating, and maintaining PFAS removal treatment technologies, including treatment media replacement and spent media destruction or disposal, as well as nontreatment actions that some PWSs may take in lieu of treatment, such as constructing new wells in an uncontaminated aquifer or interconnecting with and purchasing water from a neighboring PWS. The EPA used SafeWater MCBC to apply costs for one of the treatment technologies or nontreatment alternatives at each EP in a PWS estimated to be out of compliance with the final rule. For each affected EP, SafeWater MCBC selected from among the compliance alternatives using a decision tree procedure, described in more detail in USEPA (2024j). Next, the model estimated the cost of the chosen compliance alternative using outputs from the EPA's WBS cost estimating models. The WBS models are spreadsheet-based engineering models for individual treatment technologies, linked to a central database of component unit costs.

Specifically, the EPA used cost equations generated from the following models (USEPA, 2024m):

- the GAC WBS model (USEPA, 2024p);
- the PFAS-selective IX WBS model (USEPA, 2024q); and
- the nontreatment WBS model (USEPA, 2024r).

The *Technologies and Costs* (T&C) document (USEPA, 2024m) provides a comprehensive discussion of each of the treatment technologies, their effectiveness, and the WBS cost models as well as the equations used to calculate treatment costs. In total, there are more than 2,600 individual cost equations across the categories of capital and operation and maintenance (O&M) cost, water source, component level, flow, bed life (for GAC and IX), residuals management scenarios (for GAC and IX), and design type (for GAC). These models are available on the EPA's website at *https://www.epa.gov/sdwa/drinking-water-treatment-technology-unit-cost-models* as well as in the docket for this rule.

b. Decision Tree for Technology Selection

For EP at which baseline PFAS concentrations exceed regulatory thresholds, SafeWater MCBC selects a treatment technology or nontreatment alternative using a two-step process that both:

- Determines whether to include or exclude each alternative from consideration given the EP's characteristics and the regulatory option selected, and
- Selects from among the alternatives that remain viable based on percentage distributions derived, in part, from data on recent PWS actions in response to PFAS contamination.

Inputs to SafeWater MCBC used in Step 1 include the following:

- Influent concentrations of individual PFAS contaminants in ng/L (ppt)
- EP design flow in MGD
- TOC influent to the new treatment process in mg/L.

The EPA relied on information from the national PFAS occurrence model to inform influent PFAS concentrations. The EPA relied on *Geometries and Characteristics of Public Water Supplies* (USEPA, 2000g) and SDWIS inventory information to derive EP design flow. SafeWater MCBC selects influent TOC using the distribution shown in Table 28.

Table 28: Frequency Distribution to Estimate Influent TOC in mg/L

Percentile	Surface Water	Ground Water
0.05	0.65	0.35
0.15	1.1	0.48
0.25	1.38	0.5
0.35	1.6	0.5
0.45	1.85	0.58
0.5	1.97	0.69
0.55	2.14	0.75
0.65	2.54	1
0.75	3.04	1.39
0.85	3.63	2.01
0.95	4.81	3.8

Source: The EPA's analysis of TOC concentrations in the SYR4 ICR database.

In Step 1, SafeWater MCBC uses these inputs to determine whether to include or exclude each treatment alternative from consideration in the compliance forecast. For the treatment technologies (GAC and IX), this determination is based on estimates of each technology's performance given available data about influent water quality and the regulatory option under consideration.

The EPA assumes a small number of PWSs may be able to take nontreatment actions in lieu of treatment. The viability of nontreatment actions is likely to depend on the quantity of water being replaced because the ability to purchase from another water system is limited by the seller water system's capacity and the ability to drill another well is limited by the ability to find an accessible, sufficiently large source. Therefore, SafeWater MCBC considers nontreatment only for EP with design flows less than or equal to 3.536 MGD. The EPA estimates approximately 2 percent of systems of this size will develop new wells and approximately 6–7 percent of systems will elect to interconnect with another system to achieve compliance.

In Step 2, SafeWater MCBC selects a compliance alternative for each EP from among the alternatives that remain in consideration after Step 1. Table 29 shows the initial compliance forecast that is the starting point for this step. The percentages in Table 29 consider data presented in the T&C document (USEPA, 2024m) on actions PWSs have taken in response to PFAS contamination.

To date, the majority of PWSs for which data are available have installed GAC (USEPA, 2024m). USEPA (2024m) includes data for 52 systems, 34 of which (65%) have installed GAC. The data in USEPA (2024m) also suggest that an increasing share of PWSs have selected IX in response to PFAS since the first full-scale system treated with PFAS-selective IX in 2017. Specifically, for systems installed prior to 2017, 78% used GAC. The EPA expects this trend to continue, so the initial percentages include adjustments to account for this expectation. In addition, the performance of GAC is affected by the presence of TOC, as further described in the cost chapter of the EA (USEPA, 2024g). Accordingly, the table includes adjusted distributions for systems with higher influent TOC. Finally, while central RO/NF remains a BAT for the final rule, the EPA does not anticipate water systems will select this technology to comply with the rule, largely due to the challenges presented by managing the treatment residuals from this process.

The list of compliance alternatives in Table 29 does not include POU devices for small systems. At this time, the EPA is not including POU devices in the national cost estimates because the final rule require treatment to concentrations below the current NSF/ANSI certification standard for POU devices. However, POU treatment is reasonably anticipated to become a compliance option for small systems in the future if independent third-party certification organizations, such as NSF or ANSI develop a new certification standard that mirrors the EPA's final regulatory standard. Therefore, the decision tree excludes POU devices from consideration.

Table 29: Initial Compliance Forecast

Compliance Alternative	Design flow less than 1 MGD		Design flow 1 to less than 10 MGD		Design flow greater than or equal to 10 MGD	
	TOC less than or equal to 1.5 mg/L	TOC greater than 1.5 mg/L	TOC less than or equal to 1.5 mg/L	TOC greater than 1.5 mg/L	TOC less than or equal to 1.5 mg/L	TOC greater than 1.5 mg/L
GAC	79%	62%	81%	52%	89%	52%
PFAS-selective IX	12%	29%	11%	40%	11%	48%
Central RO/NF	0%	0%	0%	0%	0%	0%
Interconnection	7%	7%	6%	6%	0%	0%
New Wells	2%	2%	2%	2%	0%	0%

Source: The EPA's analysis of TOC concentrations in the SYR4 ICR database.

If all the compliance alternatives remain in consideration after Step 1, the decision tree uses the forecast shown in Table 29 above. If Step 1 eliminated one or more of the alternatives, SafeWater MCBC proportionally redistributes the percentages among the remaining alternatives and uses the redistributed percentages.

The EPA's approach to estimating GAC and IX performance for the final rule and all alternatives considered is discussed in detail within the cost chapter of the EA (USEPA, 2024g).

c. Work Breakdown Structure Models

The WBS models are spreadsheet-based engineering models for individual treatment technologies, linked to a central database of component unit costs. The EPA developed the WBS model approach as part of an effort to address recommendations made by the Technology Design Panel (TDP), which convened by the EPA in 1997 to review the agency's methods for estimating drinking water compliance costs (USEPA, 1997). The TDP consisted of nationally recognized drinking water experts from the EPA, water treatment consulting companies, public as well as private water utilities along with suppliers, equipment vendors, and Federal along with state regulators in addition to cost estimating professionals.

In general, the WBS approach involves breaking a process down into discrete components for the purpose of estimating unit costs. The WBS models represent improvements over past cost estimating methods by increasing comprehensiveness, flexibility, and transparency. By adopting a WBS-based approach to identify the components that should be included in a cost analysis, the models produce a more comprehensive, flexible, and transparent assessment of the capital and operating requirements for a treatment system.

Each WBS model contains the work breakdown for a particular treatment process and preprogrammed engineering criteria and equations that estimate equipment requirements for user-specified design requirements (*e.g.,* system size and influent water quality). Each model also provides unit and total cost information by component (*e.g.,* individual items of capital equipment) and totals the individual component costs to obtain a direct capital cost. Additionally, the models estimate add-on costs (*e.g.,* permits and land acquisition), indirect capital costs, and annual O&M costs, thereby producing the EPA's best estimates of complete compliance costs.

Primary inputs common to all the WBS models include design flow and average daily flow in MGD. Each WBS model has default designs (input sets) that correspond to specified categories of flow, but the models can generate designs for many other combinations of flows. To estimate costs for PFAS compliance, the EPA fit cost curves to the WBS estimates across a range of flow rates, which is described in chapter 5 of the EA (USEPA, 2024g).

Another input common to all the WBS models is ''component level'' or ''cost level.'' This input drives the selection of materials for items of equipment that can be constructed of different materials. For example, a low-cost system might include fiberglass pressure vessels and polyvinyl chloride (PVC) piping. A high-cost system might include stainless steel pressure vessels and stainless-steel piping. The component level input also drives other model assumptions that can affect the total cost of the system, such as building quality and heating and cooling. The component level input has three possible values: low cost, mid cost, and high cost. The components used in each of the estimated component/cost levels provide the treatment efficacy needed to meet the regulatory requirements. Note that the level of component (*e.g.,* plastic versus resin or stainless-steel piping and vessels) may impact the capital replacement rate but does not interfere with treatment efficacy. The EPA estimates the three levels of cost because it has found that the choice of materials associated with the installation of new treatment equipment often varies across drinking water systems. These systems may, for example, choose to balance capital cost with staff familiarity with certain materials and existing treatment infrastructure. Given this experience, the EPA models the potential variability in treatment cost based on the three component/cost levels. To estimate costs for PFAS treatment, the EPA generated separate cost equations for each of the three component levels, thus creating a range of cost estimates for use in national compliance cost estimates.

The third input common to all the WBS models is system automation, which allows the design of treatment systems that are operated manually or with varying degrees of automation (*i.e.,* with control systems that reduce the need for operator intervention). Cost equations for system automation are described in chapter 5 of the EA (USEPA, 2024g).

The WBS models generate cost estimates that include a consistent set of capital, add-on, indirect, and O&M costs. Table 30 identified these cost elements, which are common to all the WBS models and included in the cost estimates. As described and summarized in Tables 31–34 the WBS models also include technology-specific cost elements. The documentation for the WBS models provides more information on the methods and assumptions in the WBS models to estimate the costs for both the technology-specific and common cost elements (USEPA, 2024p; USEPA, 2024q; USEPA, 2024r). WBS model accuracy as well as limitations and uncertainty are described in chapter 5 of the EA (USEPA, 2024g).

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Table 30: Cost Elements Included in All WBS Models

Cost Category	Components Included
Direct Capital Costs	Technology-specific equipment (e.g., vessels, basins, pumps, treatment media, piping, valves) Instrumentation and system controls Buildings Residuals management equipment
Add-on Costs	Land Permits Pilot testing
Indirect Capital Costs	Mobilization and demobilization Architectural fees for treatment building Equipment delivery, installation, and contractor's overhead and profit Sitework Yard piping Geotechnical Standby power Electrical infrastructure Process engineering Contingency Miscellaneous allowance Legal, fiscal, and administrative Sales tax Financing during construction Construction management
O&M Costs: Technology-specific	Operator labor for technology-specific tasks (e.g., managing backwash and media replacement) Materials for O&M of technology-specific equipment Technology-specific chemical usage Replacement of technology-specific equipment that occurs on an annual basis (e.g., treatment media) Energy for operation of technology-specific equipment (e.g., mixers)
O&M Costs: Labor	Operator labor for O&M of process equipment Operator labor for building maintenance Managerial and clerical labor
O&M Costs: Materials	Materials for maintenance of booster or influent pumps Materials for building maintenance
O&M Costs: Energy	Energy for operation of booster or influent pumps Energy for lighting, ventilation, cooling, and heating
O&M Costs: Residuals	Residuals management operator labor, materials, and energy Residuals disposal and discharge costs

The GAC model can generate costs for two types of design:

- Pressure designs where the GAC bed is contained in stainless steel, carbon steel, or fiberglass pressure vessel.
- Gravity designs where the GAC bed is contained in open concrete basins.

Table 31 shows the technology-specific capital equipment and O&M requirements included in the GAC model. These items are in addition to the common WBS cost elements listed in the Table 30 above.

Table 31: Technology-Specific Cost Elements Included in the GAC Model

Cost Category	**Major Components Included**
Direct Capital Costs	Booster pumps for influent water Contactors (either pressure vessels or concrete basins) that contain the GAC bed Tanks and pumps for backwashing the contactors GAC transfer and storage equipment Spent GAC reactivation facilities (if on-site reactivation is selected) Associated piping, valves, and instrumentation
O&M Costs: Labor	Operator labor for contactor maintenance (for gravity GAC designs) Operator labor for managing backwash events Operator labor for backwash pump maintenance (if backwash occurs weekly or more frequently) Operator labor for GAC transfer and replacement
O&M Costs: Materials	Materials for contactor maintenance (accounts for vessel relining in pressure designs, because GAC can be corrosive, and for concrete and underdrain maintenance in gravity designs) Materials for backwash pump maintenance (if backwash occurs weekly or more frequently) Replacement virgin GAC (loss replacement only if reactivation is selected)
O&M Costs: Energy	Operating energy for backwash pumps
O&M Costs: Residuals	Discharge fees for spent backwash Fees for reactivating spent GAC (if off-site reactivation is selected) Labor, materials, energy, and natural gas for regeneration facility (if on-site reactivation is selected) Disposal of spent GAC (if disposal is selected)

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For small systems (less than 1 MGD) using pressure designs, the GAC model assumes the use of package treatment systems that are pre-assembled in a factory, mounted on a skid, and transported to the site. These assumptions are based on common vendor practice for these technologies, for example, see Khera et al. (2013) which says ''. . . small systems are often built as packaged, pre-engineered, or skid-mounted systems.'' The model estimates costs for package systems by costing all individual equipment line items (*e.g.,* vessels, interconnecting piping and valves, instrumentation, and system controls) in the same manner as custom-engineered systems. This approach is based on vendor practices of partially engineering these types of package plants for specific systems (*e.g.,* selecting vessel size to meet flow and treatment criteria). The model applies a variant set of design inputs and assumptions that are intended to simulate the use of a package plant and that reduce the size and cost of the treatment system. USEPA (2024p) provides complete details on the variant design assumptions used for package plants.

To generate the GAC cost equations, the EPA used the following key inputs in the GAC model:

- For pressure designs, two vessels in series with a minimum total empty bed contact time (EBCT) of 20 minutes;
- For gravity designs, contactors in parallel with a minimum total EBCT of 20 minutes; and
- Bed life varying over a range from 5,000 to 75,000 BV.

The EPA generated separate cost equations for two spent GAC management scenarios:

- Off-site reactivation under current RCRA non-hazardous waste regulations;
- Off-site disposal as a hazardous waste in a RCRA Subtitle C landfill and replacement with virgin GAC (*i.e.,* single use operation).

The T&C document (USEPA, 2024m) provides a comprehensive discussion of these and other key inputs and assumptions.

Table 32 shows the technology-specific capital equipment and O&M requirements included in the PFAS selective IX model. These items are in addition to the common WBS cost elements listed in the Table 30 above.

Table 32: Technology-Specific Cost Elements Included in the PFAS-Selective IX Model

Cost Category	Major Components Included
Direct Capital Costs	Booster pumps for influent water Pre-treatment cartridge filters Pressure vessels that contain the resin bed Tanks and pumps for initial rinse and (optionally) backwash of the resin bed Tanks (with secondary containment), pumps and mixers for delivering sodium hydroxide for use in post-treatment corrosion control (optional) Associated piping, valves, and instrumentation
O&M Costs: Labor	Operator labor for pre-treatment filters Operator labor for managing backwash/rinse events Operator labor for backwash pump maintenance (only if backwash occurs weekly or more frequently) Operator labor for resin replacement
O&M Costs: Materials	Replacement cartridges for pre-treatment filters Materials for backwash pump maintenance (only if backwash occurs weekly or more frequently) Chemical usage (if post-treatment corrosion control is selected) Replacement virgin PFAS-selective resin
O&M Costs: Energy	Operating energy for backwash/rinse pumps
O&M Costs: Residuals	Disposal of spent cartridge filters Discharge fees for spent backwash/rinse Disposal of spent resin

For small systems (less than 1 MGD), the PFAS-selective IX model assumes the use of package treatment systems that are pre-assembled in a factory, mounted on a skid, and transported to the site. The IX model estimates costs for package systems using an approach similar to that described for the GAC model, applying a variant set of inputs and assumptions that reduce the size and cost of the treatment system. USEPA (2024q) provides complete details on the variant design assumptions used for IX package plants.

To generate the IX cost equations, the EPA used the following key inputs in the PFAS-selective IX model:

- Two vessels in series with a minimum total EBCT of 6 minutes
- Bed life varying over a range from 20,000 to 260,000 BV

The EPA generated separate cost equations for two spent resin management scenarios:

- Spent resin managed as non-hazardous and sent off-site for incineration.
- Spent resin managed as hazardous and sent off-site for incineration.

The T&C document (USEPA, 2024m) provides a comprehensive discussion of these and other key inputs and assumptions.

USEPA (2024r) provides a complete description of the engineering design process used by the WBS model for nontreatment actions. The model can estimate costs for two nontreatment alternatives: interconnection with another system and drilling new wells to replace a contaminated source. Table 33 shows the technology-specific capital equipment and O&M requirements included in the model for each alternative.

Table 33: Technology-Specific Cost Elements Included in the Nontreatment Model

Cost Category	Major Components Included for Interconnection	Major Components Included for New Wells
Direct Capital Costs	Booster pumps or pressure reducing valves (depending on pressure at supply source) Concrete vaults (buried) for booster pumps or pressure reducing valves Interconnecting piping (buried) and valves	Well casing, screens, and plugs Well installation costs including drilling, development, gravel pack, and surface seals Well pumps Piping (buried) and valves to connect the new well to the system
O&M Costs: Labor	Operator labor for O&M of booster pumps or pressure reducing valves (depending on pressure at supply source) and interconnecting valves	Operator labor for operating and maintaining well pumps and valves
O&M Costs: Materials	Cost of purchased water Materials for maintaining booster pumps (if required by pressure at supply source)	Materials for maintaining well pumps
O&M Costs: Energy	Energy for operating booster pumps (if required by pressure at supply source)	Energy for operating well pumps

To generate the cost equations, the EPA used the following key inputs in the nontreatment model for interconnection:

- An interconnection distance of 10,000 feet
- Includes booster pumps designed to account for friction loss in interconnecting piping
- An average cost of purchased water of $3.35 per thousand gallons in 2022 dollars.

For new wells, the EPA used the following key inputs:

- A maximum well capacity of 500 gallons per minute (GPM), such that one new well is installed per 500 GPM of water production capacity required
- A well depth of 250 feet
- 500 feet of distance between the new wells and the distribution system.

The T&C document (USEPA, 2024m) provides a comprehensive discussion of these and other key inputs and assumptions.

d. Incremental Treatment Costs

The EPA has estimated the national level costs of the final rule associated with PFOA, PFOS, and PFHxS. As discussed in chapter 4 of the EA and detailed in the Technical Support Document for PFAS Occurrence and Contaminant Background chapter 10.1 and 10.3, there are limitations with nationally representative occurrence information for the other contaminants in the final rule (PFNA, HFPO–DA and PFBS). Specifically, HFPO–DA does not currently have a completed nationally representative dataset while PFNA and PFBS were not included in the national occurrence model because of limited results reported above the minimum reporting levels in UCMR 3. As described in the Technical Support Document for PFAS Occurrence and Contaminant Background chapter 10.3.2, non-targeted state monitoring datasets were used for extrapolation of PFNA, HFPO–DA, and PFBS in lieu of a nationally representative dataset. The EPA used conservative assumptions in this extrapolation to generate conservative cost estimates. As demonstrated in this analysis, the Hazard Index, PFNA, and HFPO–DA MCLs meaningfully increase public health protection at modest additional costs. Because of the increased uncertainty associated with PFNA, HFPO–DA and PFBS, the additional treatment cost from co-occurrence of PFNA, HFPO–DA, PFBS at systems already required to treat because of PFOA, PFOS, or PFHxS MCL and Hazard Index exceedances are not quantitatively assessed in the national cost estimates. These three PFAS' treatment costs are summarized here in this section and detailed in appendix N.3 of the EA (USEPA, 2024e). Likewise, treatment costs for systems that exceed the Hazard Index based on the combined occurrence of PFNA, HFPO–DA, PFBS, and PFHxS (where PFHxS itself does not exceed its HBWC of 10 ng/L) are not included in the national monetized cost estimates and are also summarized in this section and detailed in appendix N.3 of the EA (USEPA, 2024e).

In the EA for the proposed PFAS NPDWR, the EPA used a model system approach to illustrate the potential incremental costs for removing PFAS not included in the national economic model. After considering public comments on the incremental cost analysis, the EPA decided to further explore the incremental costs associated with the Hazard Index and MCLs with a national level sensitivity analysis for the final rule.

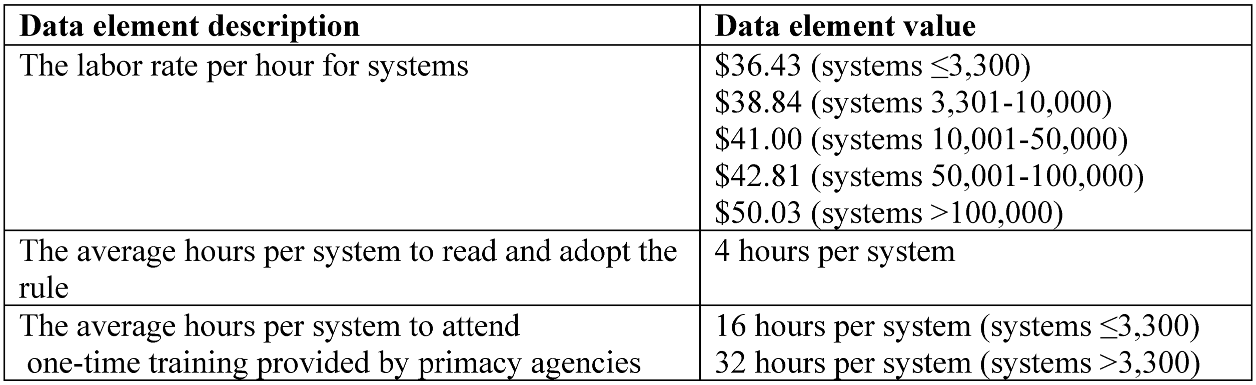
When the modeled occurrence data for PFNA, HFPO–DA, PFBS is incorporated into the SafeWater MCBC model, the estimated number of EP exceeding one or more MCLs, and therefore required to treat or use a different water source, increases to 9,471 from 9,043. This results in an increase in the expected national costs. Under the primary analyses, the expected total national cost is $1,549 million over the EPA's period of analysis (2024–2105) for the PFOA, PFOS, and PFHxS MCLs. When considering the additional incremental national cost impacts of the Hazard Index MCL for, PFNA, HFPO–DA, and PFBS (and individual MCLs for PFNA and HFPO–DA) the expected national costs of the final rule increase to $1,631 million at, or approximately a 5 percent national cost increase.

For further detail on the assumptions and findings of the EPA's analysis of incremental costs of other PFAS, see appendix N.3 and section XII.A of this preamble.

e. PWS Implementation Administration Costs

The EPA estimated PWS costs associated with one-time actions to begin implementation of the rule including reading and understanding the rule and attending training provided by primacy agencies. The average unit costs for PWSs are based on the following burden assumptions: (1) The EPA anticipates that the majority of water systems will likely not read the entirety of the rule preamble (as they are not required to do so) but focus their time and attention on understanding the regulatory requirements through the CFR regulatory text, relevant portions of the preamble, the EPA provided fact sheets and small system guidance documents, and state provided summaries documents; (2) Additionally, the EPA anticipates that system staff will attend primacy agency PFAS rule trainings to reenforce the systems' understanding of the final rule. The EPA assumes that systems will conduct these activities during years one through three of the analysis period. Table 34 lists the data elements and corresponding values associated with calculating the costs of these one-time implementation administration actions.

Table 34: Implementation Administration Startup Costs ($2022)



Data element description	Data element value
The labor rate per hour for systems	$36.43 (systems ≤3,300) $38.84 (systems 3,301-10,000) $41.00 (systems 10,001-50,000) $42.81 (systems 50,001-100,000) $50.03 (systems >100,000)
The average hours per system to read and adopt the rule	4 hours per system
The average hours per system to attend one-time training provided by primacy agencies	16 hours per system (systems ≤3,300) 32 hours per system (systems >3,300)

Estimated national annualized PWS implementation and administration startup costs for the final rule are $1.33 million. National annualized PWS cost estimates are further summarized in Table 39.

f. PWS Monitoring Costs

The final rule requires initial and long-term monitoring. As Table 35 shows, surface and ground water systems serving greater than 10,000 people will collect one sample each quarter, at each EP, during the initial 12-month monitoring period. Surface water systems serving 10,000 or fewer people are also required to collect a quarterly sample at each EP during the initial 12-month period. Ground water systems that serve 10,000 or fewer people will be required to sample once at each EP on a semi-annual basis for the first 12-month monitoring period.

Long-term monitoring schedules are based on specific EP sampling results (*i.e.,* water systems can have different EP within the system on different monitoring schedules). Long-term monitoring requirements differ based on whether a system can demonstrate during the initial monitoring period or once conducting long-term monitoring that an EP is below the trigger levels for regulated PFAS. The trigger levels are set as one-half the MCLs: 2.0 ng/L for PFOA and PFOS, 5 ng/L each for PFHxS, PFNA, and HFPO–DA, and 0.5 for the Hazard Index. EP below the trigger level values during the initial 12-month monitoring period and in future long-term monitoring periods may conduct triennial monitoring and collect one triennial sample at that EP. For EP with concentration values at or above a trigger level, a quarterly sample must be taken at that EP following initial monitoring. EP that demonstrate they are "reliably and consistently"[21] below the MCLs following four consecutive quarterly samples are eligible to conduct annual monitoring. After three annual samples at that EP showing no results at or above a trigger level, the location can further reduce to triennial monitoring.

For any samples that are above detection, the system will analyze the FRB samples collected at the same time as the monitoring sample. Systems that have an MCL exceedance will collect one additional sample from the relevant EP to confirm the results.

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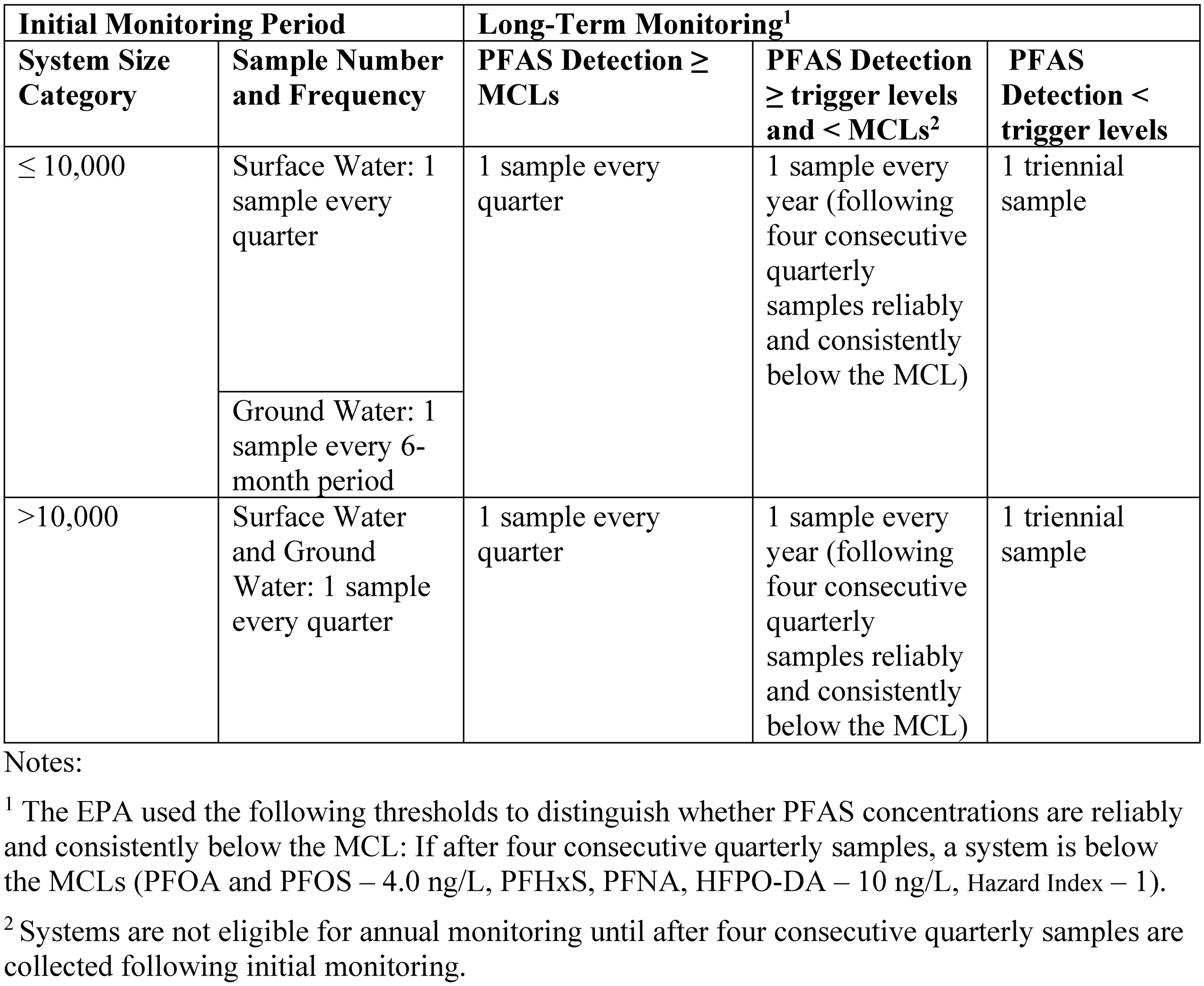
[21] The definition of reliably and consistently below the MCL means that each of the samples contains regulated PFAS concentrations below the applicable MCLs. For the PFAS NPDWR, this demonstration of reliably and consistently below the MCL would include consideration of at least four quarterly samples at an EP below the MCL, but states will make their own determination as to whether the detected concentrations are reliably and consistently below the MCL.

Table 35: Modeled Initial and Long-Term Sampling Frequencies Per System EP



Initial Monitoring Period		Long-Term Monitoring[1]		
System Size Category	**Sample Number and Frequency**	**PFAS Detection ≥ MCLs**	**PFAS Detection ≥ trigger levels and < MCLs[2]**	**PFAS Detection < trigger levels**
≤ 10,000	Surface Water: 1 sample every quarter	1 sample every quarter	1 sample every year (following four consecutive quarterly samples reliably and consistently below the MCL)	1 triennial sample
	Ground Water: 1 sample every 6-month period			
>10,000	Surface Water and Ground Water: 1 sample every quarter	1 sample every quarter	1 sample every year (following four consecutive quarterly samples reliably and consistently below the MCL)	1 triennial sample

Notes:

[1] The EPA used the following thresholds to distinguish whether PFAS concentrations are reliably and consistently below the MCL: If after four consecutive quarterly samples, a system is below the MCLs (PFOA and PFOS – 4.0 ng/L, PFHxS, PFNA, HFPO-DA – 10 ng/L, Hazard Index – 1).

[2] Systems are not eligible for annual monitoring until after four consecutive quarterly samples are collected following initial monitoring.

For the national cost analysis, the EPA assumes that systems with either UCMR 5 data or monitoring data in the State PFAS Database (see section 3.1.4 in USEPA, 2024g) will not conduct the initial year of monitoring as allowed by the final rule. As a simplifying assumption for the cost analysis, the EPA assumes all systems serving a population of greater than 3,300 have UCMR 5 data and those with 3,300 or less do not. For the State PFAS Database, the EPA relied on the PWSIDs stored in the database and exempted those systems from the first year of monitoring in the cost analysis. Note these simplifying assumptions may result in a small underestimate of initial monitoring costs. Under UCMR 5, individual water systems would be able to request the full release of data from the labs for use in determining their compliance monitoring frequency. PWSs may be able to use these lab analyses to demonstrate a ''below trigger level'' concentration using the UCMR 5 analyses by following up with the lab for a more detailed results report.

The EPA used system-level distributions of PFOA, PFOS, and PFHxS, as described in Cadwallader et al. (2022), to simulate EP concentrations and estimate PFAS occurrence relative to the final rule MCLs and trigger levels. Based on these occurrence distributions, the EPA estimates that the large majority of water systems subject to the rule (approx. 52,000–57,000) will have EP with concentrations below the trigger levels and would conduct reduced monitoring on a triennial basis. The EPA estimates that the remainder of water systems subject to the rule (approx. 9,000–15,000) will have at least one or more EP exceed the trigger level and therefore would be required to conduct quarterly monitoring.

The EPA assumes that systems with an MCL exceedance will implement actions to comply with the MCL by the compliance date. The EPA assumes a treatment target,[22] for systems required to treat for PFAS, that includes a margin of safety so finished water PFAS levels at these systems are 80 percent of the MCLs. In the final rule, in order to reduce burden associated with monitoring, the EPA is adding an annual tier of sampling for any system with concentrations ''reliably and consistently'' [23] below the MCL but not consistently below the trigger level. The EPA believes this tier would likely

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[22] A treatment target is a contaminant concentration that a PWS has designed and operated their water system to meet. The EPA assumes all PWS will target 80% of the MCLs.

[23] The definition of reliably and consistently below the MCL means that each of the samples contains regulated PFAS concentrations below the applicable MCLs. For the PFAS NPDWR, this demonstration of reliably and consistently below the MCL would include consideration of at least four quarterly samples at an EP below the MCL, but states will make their own determination as to whether the detected concentrations are reliably and consistently below the MCL.

apply to most systems treating their water for regulated PFAS, at least for the first three years of treatment. Therefore, in the model, the EPA assumes EP that have installed treatment will take one year of quarterly samples, then continue to sample on an annual basis after that. The final rule allows EP showing no results at or above a trigger level after three annual samples to further reduce to triennial monitoring. In the national cost analysis, the EPA does not model this possibility nor does the EPA model instances where water systems are triggered back into quarterly monitoring after installing treatment.

For all systems, the activities associated with the sample collection in the initial 12-month monitoring period are the labor burden and cost for the sample collection and analysis, as well as a review of the sample results. Table 36 presents the data elements and corresponding values associated with calculating sampling costs during the implementation monitoring period.

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Table 36: Sampling Costs ($2022)

Data Element Description	Data Element Value
The labor rate per hour for systems	$36.43 (systems ≤3,300) $38.84 (systems 3,301-10,000) $41.00 (systems 10,001-50,000) $42.81 (systems 50,001-100,000) $50.03 (systems >100,000)
The number of samples per EP per monitoring round for the initial monitoring in Year 1	2 samples (Ground Water systems ≤10,000) 4 samples (all systems) [1]
The number of samples per EP per long-term monitoring year for EPs that equal or exceed the MCLs	4 samples per year
The number of samples per EP per long-term monitoring year for EP < the MCLs and ≥ the trigger levels[2]	1 sample per year, following 4 quarterly samples reliably and consistently below the MCLs
The number of samples per EP per long-term monitoring round for EP < the trigger levels	1 sample every three years
The hours per sample to travel to sampling locations, collect samples, record any additional information, submit samples to a laboratory, and review results	1 hour
The laboratory analysis cost per sample for EPA Method 537.1	$309
The laboratory analysis cost per sample for the FRB under EPA Method 537.1	$273 [3]

Notes:

[1] Systems greater than 3,300 will rely on UCMR 5 data and a subset of other systems will rely on data in the State PFAS Monitoring Database discussed in USEPA, 2024g.

[2] The EPA used the following thresholds to distinguish whether PFAS concentrations are reliably and consistently below the MCL: If after four consecutive quarterly samples, a system is below the MCLs (PFOA and PFOS – 4.0 ng/L, PFHxS, PFNA, HFPO-DA – 10 ng/L, Hazard Index – 1).

[3] This incremental sample cost applies to all samples that exceed MDLs. The EPA used the Method 537.1 detection limits to apply this cost because Method 533 does not include detection limits.

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Estimated national annualized PWS sampling costs for the final rule have an expected value of $36.23 million. National annualized PWS cost estimates are further summarized in Table 39.

g. Treatment Administration Costs

Any system with an MCL exceedance adopts either a treatment or nontreatment alternative to comply with the rule. The majority of systems are anticipated to install treatment technologies while a subset of systems will choose alternative methods. The EPA assumes that systems will bear administrative costs associated with these treatment or nontreatment compliance actions (*i.e.,* permitting costs). The EPA assumes that systems will install treatment in the fifth year of the period of analysis. In addition, after installation of treatment, the EPA assumes that systems will spend an additional 2 hours per treating EP compiling data for and reviewing treatment efficacy with their primacy agency during their triennial sanitary survey. Table 37 presents the data elements and corresponding values associated with calculating treatment administration costs.

Table 37: Treatment Administration Costs ($2022)

Data element description	**Data element value**
The labor rate per hour for systems	$36.43 (systems ≤3,300) $38.84 (systems 3,301-10,000) $41.00 (systems 10,001-50,000) $42.82 (systems 50,001-100,000) $50.03 (systems >100,000)
The hours per EP for a system to notify, consult, and submit a permit request for treatment installation [a]	3 hours (systems ≤100) 5 hours (systems 101-500) 7 hours (systems 501-1,000) 12 hours (systems 1,001-3,300) 22 hours (systems 3,301-50,000) 42 hours (systems >50,000)
The additional hours per EP the system will spend every 3 years during a sanitary survey after PFAS related treatment is installed	2 hours, at EP that have installed treatment
The hours per EP for a system to notify, consult, and submit a permit request for source water change or alternative method[1]	6 hours

Notes:

[1] The EPA applied the cost per EP for this EA because the notification, consultation, and permitting process occurs for individual EP.

h. Public Notification (PN) Costs

The EPA's cost analysis assumes full compliance with the rule throughout the period of analysis and, as a result, the EPA does not estimate costs for the PN requirements in the final rule for systems with certain violations. The final rule designates MCL violations for PFAS as Tier 2, which requires systems to provide PN as soon as practical, but no later than 30 days after the system learns of the violation. The system must repeat notice every three months if the violation or situation persists unless the primacy agency determines otherwise. At a minimum, systems must give repeat notice at least once per year. The final rule also designates monitoring and testing procedure violations as Tier 3, which requires systems to provide public notice no later than one year after the system learns of the violation. The system must repeat the notice annually for as long as the violation persists. CWSs may deliver Tier 3 PNs in their CCR if the timing, content, and delivery requirements are met according to 40 CFR 141.204(d). Using the CCR to deliver Tier 3 PNs can minimize the burden on systems by reducing delivery costs. For approximate estimates of the potential burden associated with Tier 2 and 3 PNs, please see USEPA (2024g).

i. Primacy Agency Costs

The EPA assumes that primacy agencies will have upfront implementation costs as well as costs associated with system actions related to sampling and treatment. The activities that primacy agencies are expected to carry out under the final rule include:

- Reading and understanding the rule, providing internal primacy agency officials training for the rule implementation, updating sanitary survey standard operating procedures,
- Primacy package application, including making regulatory changes to the Federal rule where applicable,
- Providing systems with training and technical assistance during the rule implementation,
- Reporting to the EPA on an ongoing basis any PFAS-specific information under 40 CFR 142.15 regarding violations as well as enforcement actions and general operations of PWS programs,
- Performing inspection of PFAS related treatment during sanitary surveys every three years

- Reviewing the sample results during the implementation monitoring period and the SMF period, and
- Reviewing and consulting with systems on the installation of treatment technology or alternative methods, including source water change.

For the last three activities listed above, the primary agency burdens are incurred in response to action taken by PWSs; for instance, the cost to primacy agencies of reviewing sample results depends on the number of samples taken at each EP by each system under an agency's jurisdiction. Table 38 presents the data elements and corresponding values associated with calculating primacy agency costs.

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Table 38: Primacy Agency Costs ($2022)

Data element description	Data element value
The labor rate per hour for primacy agencies[1]	$59.69
The average hours per primacy agency to read and understand the rule, update sanitary survey standard operating procedures, and train internal staff.	4,020 hours per primacy agency
The average hours for a primacy agency to develop and adopt state-level regulations	300 hours per primacy agency
The average hours per primacy agency to provide initial training and technical assistance to systems	1,500 hours per primacy agency
The average hours per primacy agency to report annually to the EPA information under 40 CFR 142.15 regarding violations, variances and exemptions, enforcement actions and general operations of state PWS programs[2]	0
The hours per sample for a primacy agency to review sample results	1 hour
The hours per EP for a primacy agency to review and consult on installation of a treatment technology	80 hours (systems serving ≤3,300) 70 hours (systems serving 3,301 to 50,000) 50 hours (systems serving >50,000)
The additional hours per EP the primacy agency will spend every 3 years after PFAS-related treatment is installed during a sanitary survey	2 hours per EP that installs treatment every 3 years post installation
The hours per EP for a primacy agency to review and consult on a source water change	4 hours

Notes:

[1] In USBLS (2022), state employee wage rate of $33.91 from National Occupational Employment and Wage Estimates, United States, BLS SOC Code 19-2041, "State Government, excluding schools and hospitals - Environmental Scientists and Specialists, Including Health," hourly mean wage rate. May 2020 data (published in March 2021): https://www.bls.gov/oes/current/oes192041.htm. Wages are loaded using a factor of 62.2 from the Bureau of Labor Statistics (BLS) Employer Costs for Employee Compensation report, Table 3, March 2020. Percent of total compensation - Wages and Salaries - All Workers - State and Local Government Workers (https://www.bls.gov/news.release/archives/ecec_06182020.pdf). See worksheet BLS Table 3. The final loaded wage is adjusted for inflation.

[2] The EPA assumes that the final PFAS rule will have no discernable incremental burden for quarterly or annual reports to SDWIS Fed.

Estimated national annualized primacy agency costs for the final rule have an expected value of $4.65 million. National annualized cost estimates are further summarized in Table 39.

In addition to the costs described above, a primacy agency may also have to review the certification of any Tier 2 or 3 PNs sent out by systems. The EPA assumes full compliance with the final rule and therefore does not include this cost in national estimated cost totals but provides a brief discussion of the possible primacy agency burden associated with this component in USEPA (2024g).

In Table 39, the EPA summarizes the total annualized quantified cost of the final rule at a 2 percent discount rate expressed in millions of 2022 dollars. The first three rows show the annualized PWS sampling costs, the annualized PWS implementation and administrative costs, and the annualized PWS treatment costs. The fourth row shows the sum of the annualized PWS costs. The expected annualized PWS costs are $1,544 million. The uncertainty range for annualized PWS costs are $1,431 million to $1,667 million. Finally, annualized primacy agency implementation and administrative costs are added to the annualized PWS costs to calculate the total annualized cost of the final rule. The expected total annualized cost of the final rule is $1,549 million. The uncertainty range for the total annualized costs of the final rule is $1,436 million to $1,672 million. The EPA notes that treatment costs associated with the rule are the most significant contribution to overall rule costs for the final rule and the regulatory alternatives.

Table 39: National Annualized Costs, Final Rule (PFOA and PFOS MCLs of 4.0 ng/L each, PFHxS, PFNA, and HFPO-DA MCLs of 10 ng/L each, and Hazard Index of 1) (Million $2022)

	2% Discount Rate		
	5th Percentile[1]	**Expected Value**	**95th Percentile[1]**
Annualized PWS Sampling Costs	$33.63	$36.23	$39.03
Annualized PWS Implementation and Administration Costs	$1.33	$1.33	$1.33
Annualized PWS Treatment Costs	$1,395.23	$1,506.44	$1,627.65
Total Annualized PWS Costs[2,3,4]	$1,431.00	$1,544.00	$1,667.10
Primacy Agency Rule Implementation and Administration Cost	$4.35	$4.65	$4.97
Total Annualized Rule Costs[2,3,4]	$1,435.70	$1,548.64	$1,672.10

Notes:

Detail may not add exactly to total due to independent rounding. 5th and 95th percentile values for total rule costs are not additive across cost categories as the categories are not completely correlated.

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.J of this preamble and Table 74. This range does not include the uncertainty described in Table 43.

[2] The national level cost estimates for PFHxS are reflective of both the total national cost for PFHxS individual MCL exceedances, and Hazard Index MCL exceedances where PFHxS is present above its HBWC while one or more other Hazard Index PFAS is also present in that same mixture. Total quantified national cost values do not include the incremental treatment costs associated with the co-occurrence of PFNA, HFPO-DA, and PFBS. The EPA has considered the additional national costs of the Hazard Index and individual MCLs associated with HFPO-DA, PFBS, and PFNA occurrence in a quantified sensitivity analysis; see appendix N.3 of the EA (USEPA, 2024e) for the analysis and more information.

[3] PFAS-contaminated wastes are not considered RCRA regulatory or characteristic hazardous wastes at this time and therefore total costs reported in this table do not include costs associated with hazardous waste disposal of spent filtration materials. To address stakeholder concerns about potential costs for disposing PFAS-contaminated wastes as hazardous should they be regulated as such in the future, the EPA conducted a sensitivity analysis with an assumption of hazardous waste disposal for illustrative purposes only. See appendix N.2 of the EA (USEPA, 2024e) for additional detail.

[4] See Table 72 for a list of the nonquantifiable costs, and the potential direction of impact these costs would have on the estimated monetized total annualized costs in this table.

In Tables 40, 41, and 42, the EPA summarizes the total annualized quantified cost of options 1a, 1b, and 1c, respectively.

Table 40: National Annualized Costs, Option 1a (PFOA and PFOS MCLs of 4.0 ng/L; Million $2022)

	2% Discount Rate		
	5th Percentile[1]	**Expected Value**	**95th Percentile[1]**
Annualized PWS Sampling Costs	$33.37	$35.98	$38.77
Annualized PWS Implementation and Administration Costs	$1.33	$1.33	$1.33
Annualized PWS Treatment Costs	$1,383.33	$1,495.14	$1,616.15
Total Annualized PWS Costs[2,3]	$1,419.20	$1,532.44	$1,654.80
Primacy Agency Rule Implementation and Administration Cost	$4.34	$4.63	$4.95
Total Annualized Rule Costs[2,3]	$1,423.60	$1,537.07	$1,660.30

Notes:

Detail may not add exactly to total due to independent rounding. 5th and 95th percentile values for total rule costs are not additive across cost categories as the categories are not completely correlated.

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.J of this preamble and Table 74. This range does not include the uncertainty described in Table 43.

[2] PFAS-contaminated wastes are not considered RCRA regulatory or characteristic hazardous wastes at this time and therefore total costs reported in this table do not include costs associated with hazardous waste disposal of spent filtration materials. To address stakeholder concerns about potential costs for disposing PFAS-contaminated wastes as hazardous should they be regulated as such in the future, the EPA conducted a sensitivity analysis with an assumption of hazardous waste disposal for illustrative purposes only. See appendix N.2 of the EA (USEPA, 2024e) for additional detail.

[3] See Table 72 for a list of the nonquantifiable costs, and the potential direction of impact these costs would have on the estimated monetized total annualized costs in this table.

Table 41: National Annualized Costs, Option 1b (PFOA and PFOS MCLs of 5.0 ng/L; Million $2022)

	2% Discount Rate		
	5th Percentile[1]	**Expected Value**	**95th Percentile[1]**
Annualized PWS Sampling Costs	$31.07	$33.29	$35.71
Annualized PWS Implementation and Administration Costs	$1.33	$1.33	$1.33
Annualized PWS Treatment Costs	$1,065.30	$1,153.31	$1,250.22
Total Annualized PWS Costs[2,3]	$1,098.40	$1,187.92	$1,286.50
Primacy Agency Rule Implementation and Administration Cost	$3.98	$4.21	$4.47
Total Annualized Rule Costs[2,3]	$1,102.60	$1,192.13	$1,291.40

Notes:

Detail may not add exactly to total due to independent rounding. 5th and 95th percentile values for total rule costs are not additive across cost categories as the categories are not completely correlated.

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.J of this preamble and Table 74. This range does not include the uncertainty described in Table 4f3.

[2] PFAS-contaminated wastes are not considered RCRA regulatory or characteristic hazardous wastes at this time and therefore total costs reported in this table do not include costs associated with hazardous waste disposal of spent filtration materials. To address stakeholder concerns about potential costs for disposing PFAS-contaminated wastes as hazardous should they be regulated as such in the future, the EPA conducted a sensitivity analysis with an assumption of hazardous waste disposal for illustrative purposes only. See appendix N.2 of the EA (USEPA, 2024e) for additional detail.

[3] See Table 72 for a list of the nonquantifiable costs, and the potential direction of impact these costs would have on the estimated monetized total annualized costs in this table.

Table 42: National Annualized Costs, Option 1c (PFOA and PFOS MCLs of 10.0 ng/L; Million $2022)

	2% Discount Rate		
	5th Percentile[1]	**Expected Value**	**95th Percentile[1]**
Annualized PWS Sampling Costs	$26.11	$27.48	$28.97
Annualized PWS Implementation and Administration Costs	$1.33	$1.33	$1.33
Annualized PWS Treatment Costs	$431.37	$467.12	$507.50
Total Annualized PWS Costs[2,3]	$459.50	$495.93	$537.21
Primacy Agency Rule Implementation and Administration Cost	$3.27	$3.37	$3.48
Total Annualized Rule Costs[2,3]	$462.87	$499.29	$540.68

Notes:

Detail may not add exactly to total due to independent rounding. 5th and 95th percentile values for total rule costs are not additive across cost categories as the categories are not completely correlated.

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.J of this preamble and Table 74. This range does not include the uncertainty described in Table 43.

[2] PFAS- contaminated wastes are not considered RCRA regulatory or characteristic hazardous wastes at this time and therefore total costs reported in this table do not include costs associated with hazardous waste disposal of spent filtration materials. To address stakeholder concerns about potential costs for disposing PFAS-contaminated wastes as hazardous should they be regulated as such in the future, the EPA conducted a sensitivity analysis with an assumption of hazardous waste disposal for illustrative purposes only. See appendix N.2 of the EA (USEPA, 2024e) for additional detail.

[3] See Table 72 for a list of the nonquantifiable costs, and the potential direction of impact these costs would have on the estimated monetized total annualized costs in this table.

j. Data Limitations and Uncertainties in the Cost Analysis

Table 43 lists data limitations and characterizes the impact on the quantitative cost analysis. The EPA notes that in most cases it is not possible to judge the extent to which a particular limitation or uncertainty could affect the cost analysis. The EPA provides the potential direction of the impact on the cost estimates when possible but does not prioritize the entries with respect to the impact magnitude.

Table 43: Limitations that Apply to the Cost Analysis for the Final PFAS Rule

Uncertainty/ Assumption	Effect on Quantitative Analysis	Notes
WBS engineering cost model assumptions and component costs	Uncertain	The WBS engineering cost models require many design and operating assumptions to estimate treatment process equipment and operating needs. Chapter 5 of the EA (USEPA, 2024g) addressed the bed life assumption. The *Technologies and Costs* document (USEPA, 2024m) and individual WBS models in the rule docket provide additional information. The component-level costs approximate national average costs, which can over- or under-estimate costs at systems affected by the final rule.
Compliance forecast	Uncertain	The forecast probabilities are based on historical full-scale compliance actions. Site-specific water quality conditions, changes in technology, and changes in market conditions can result in future technology selections that differ from the compliance forecast.
TOC concentration	Uncertain	The randomly assigned values from the two national distributions are based on a limited dataset. Actual TOC concentrations at systems affected by the final rule can be higher or lower than the assigned values.
Insufficient UCMR 3 data for PFBS and PFNA and no UCMR 3 data for HFPO-DA were available to incorporate into the Bayesian hierarchical occurrence model	Underestimate	The final rule regulates PFNA, HFPO-DA, and PFBS in addition to the PFAS modeled in the primary analysis. In instances when concentrations of PFBS, PFNA, and/or HFPO-DA are high enough to cause or contribute to Hazard Index exceedances or PFNA and/or HFPO-DA are high enough to cause individual MCL exceedances, the modeled costs in the primary analysis may be underestimated. If these PFAS occur in isolation at levels that affect treatment decisions, or if they occur in sufficient concentration to result in an exceedance when the concentration of PFHxS alone would be below the HBWC, then costs would be underestimated. Note that the EPA has conducted a sensitivity analysis of and considered the potential changes in treatment cost associated with the occurrence of PFNA, HFPO-DA, and PFBS using which is discussed in detail in appendix N.3 of the EA (USEPA, 2024e).

Uncertainty/ Assumption	**Effect on Quantitative Analysis**	**Notes**
POU not included in compliance forecast	Overestimate	If POU devices can be certified to meet concentrations that satisfy the final rule, then small systems may be able to reduce costs by using a POU compliance option instead of centralized treatment or source water changes.
Process wastes not classified as hazardous	Underestimate	The national cost analysis reflects the assumption that PFAS-contaminated wastes are not considered RCRA regulatory or characteristic hazardous wastes. To address stakeholder concerns, including those raised during the SBREFA process, the EPA conducted a sensitivity analysis with an assumption of hazardous waste disposal for illustrative purposes only. As part of this analysis, the EPA generated a second full set of unit cost curves that are identical to the curves used for the national cost analysis with the exception that spent GAC and spent IX resin are considered hazardous. If in the future PFAS-contaminated wastes require handling as hazardous wastes, the residuals management costs in the WBS treatment cost models are expected to be higher. See appendix N.2 of the EA (USEPA, 2024e) for a sensitivity analysis describing the potential increase in costs associated with hazardous waste disposal at 100 percent of systems treating for PFAS. The costs estimated in appendix N are consistent with the EPA OLEM's *Interim Guidance on the Destruction and Disposal of Perfluoroalkyl and Polyfluoroalkyl Substances and Materials Containing Perfluoroalkyl and Polyfluoroalkyl Substances* (USEPA, 2020d) and subsequent updates.
Population served held constant over time.	Uncertain	All PWS populations served were held constant over the period of analysis as not all locations have reliable information on population changes over time. If population served by affected PWSs increases (or decreases), then the estimated costs are likely underestimated (or overestimated).

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E. Nonquantifiable costs of the final rule

As described in section j. (*Data Limitations and Uncertainties in the Cost Analysis*) above, given the available occurrence data for the other compounds in the rule (PFNA, HFPO–DA, and PFBS) and the regulatory thresholds under consideration, the EPA considered national costs associated with potential Hazard Index exceedances as a direct result of these compounds in a sensitivity analysis; therefore, the additional treatment cost, from co-occurrence of PFNA, HFPO–DA, PFBS, at systems already required to treat because of PFOA, PFOS, or PFHxS MCL and Hazard Index exceedances are not presented in the national cost estimates above. Nor are treatment costs for systems that exceed the Hazard Index based on the combined occurrence of PFHxS (where PFHxS itself does not exceed 10 ng/L), PFNA, HFPO–DA, and PFBS presented in the national monetized cost estimates above. Treatment costs for the individual PFNA and HFPO–DA MCLs are also not considered above. For further discussion of how the EPA considered the costs of the five individual MCLs and the HI MCL, see section XII.A.4 of this preamble. These potential additional costs are described in greater detail in section 5.3.1.4 of USEPA (2024g) and appendix N.3 of USEPA (2024e). When considering the national cost impacts of the Hazard Index MCL for PFNA, HFPO–DA, and PFBS (and individual MCLs for PFNA and HFPO–DA) the expected national costs increase from $1,549 million to $1,631 million, or approximately a 5 percent national cost increase.

PFAS-contaminated wastes are not considered RCRA regulatory or characteristic hazardous wastes at this time and therefore total costs reported in this table do not include costs associated with hazardous waste disposal of spent filtration materials. To address stakeholder concerns, including those raised during the Small Business Regulatory Enforcement Fairness Act (SBREFA) process, the EPA conducted a sensitivity analysis with an assumption of hazardous waste disposal for illustrative purposes only. As part of this analysis, the EPA generated a second full set of unit cost curves that are identical to the curves used for the national cost analysis with the exception that spent GAC and spent IX resin are considered hazardous. If in the future PFAS-contaminated wastes require handling as hazardous wastes, the residuals management costs are expected to be higher. See appendix N.2 of the EA for a sensitivity analysis describing the potential increase in costs associated with hazardous waste disposal (USEPA, 2024e).

F. Method for Estimating Benefits

The EPA's quantification of health benefits resulting from reduced PFAS exposure in drinking water was driven by PFAS occurrence estimates, PK model availability, information on exposure-response relationships, and economic data to monetize the impacts. In the EA, the EPA either quantitatively assesses or qualitatively discusses health endpoints associated with exposure to PFAS. The EPA assesses potential benefits quantitatively if there is evidence of an association between PFAS exposure and health effects, if it is possible to link the outcome to risk of a health effect, and if there is no overlap in effect with another quantified endpoint in the same outcome group. Particularly, the most consistent epidemiological associations with PFOA and PFOS include decreased immune system response, decreased birthweight, increased serum lipids, and increased serum liver enzymes (particularly alanine transaminase, ALT). The available evidence indicates effects across immune, developmental, cardiovascular, and hepatic organ systems at the same or approximately the same level of exposure.

Table 44 presents an overview of the categories of health benefits expected to result from the implementation of treatment that reduces PFAS levels in drinking water. Of the PFAS compounds included in the final rule, the EPA quantifies some of the adverse health effects associated with PFOA and PFOS. These compounds have likely evidence linking exposure to a particular health endpoint and have reliable PK models connecting the compound to PFAS blood serum. PK models are tools for quantifying the relationship between external measures of exposure and internal measures of dose. Benefits from avoided adverse health effects of PFHxS, PFNA, HFPO–DA, and PFBS are discussed qualitatively in this section.

As Table 44 demonstrates, only a subset of the potential health effects of reduced PFAS in drinking water can be quantified and monetized. The monetized benefits evaluated in the EA for the final rule include changes in human health risks associated with CVD and infant birth weight from reduced exposure to PFOA and PFOS in drinking water and RCC from reduced exposure to PFOA. The EPA also quantified benefits from reducing bladder cancer risk due to the co-removal of non-PFAS pollutants via the installation of drinking water treatment, discussed in greater detail in USEPA (2024g). The EPA quantified benefits associated with PFOS effects on liver cancer and PFNA effects on birth weight in sensitivity analyses.

The EPA was not able to quantify or monetize other benefits, including those related to other reported health effects including immune, liver, endocrine, metabolic, reproductive, musculoskeletal, or other cancers. The EPA discusses these benefits qualitatively in more detail in this section, as well as in section 6.2 of USEPA (2024g).

Table 44: Overview of Health Benefits Categories Considered in the Analysis of Changes in PFAS Drinking Water Levels

Health Outcome		PFAS Compound[a,b,d]		Benefits Analysis	
Category	**Endpoint**	**PFOA**	**PFOS**	**Discussed Quantitatively**	**Discussed Qualitatively**
Lipids	Total cholesterol (TC)	X	X	X	
	High-density lipoprotein cholesterol (HDLC)	X[c]	X[c]	X	
	Low-density lipoprotein cholesterol (LDLC)	X	X		X
CVD	Blood pressure (BP)		X	X	
Developmental	Birth weight	X	X	X	
	Small for gestational age (SGA), non-birth weight developmental	X			X
Hepatic	Alanine transaminase (ALT)	X	X		X
Immune	Antibody response (tetanus, diphtheria)	X	X		X
Metabolic	Leptin	X			X
Musculoskeletal	Osteoarthritis, bone mineral density	X			X
Cancer	Renal Cell Carcinoma (RCC)	X		X	
	Liver		X	X[e]	
	Testicular	X			X

Notes:

[a] Fields marked with "X" indicate the PFAS compound for which there is evidence of an association with a given health outcome in humans.

[b] Outcomes with indicative evidence of an association between a PFAS compound and a health outcome are assessed quantitatively unless (1) there is an overlap within the same outcome group (e.g., LDLC overlaps with TC and SGA overlaps with low birth weight), or (2) it is not possible to link the outcome to the risk of the health effect (e.g., evidence is inconclusive regarding the relationship between PFOS exposure, leptin levels and associated health outcomes). Such health outcomes are discussed qualitatively.

[c] Although evidence of associations between HDLC and PFOA and PFOS was mixed, certain individual studies reported robust associations in general adult populations. Based on comments and recommendations from the EPA SAB, the EPA assessed HDLC in a sensitivity analysis.

[d] Note that only PFOA and PFOS effects were modeled in the assessment of benefits under the final rule. For another PFAS in the rule, PFNA, the best available finalized analysis is based on studies published before 2018 (ATSDR, 2021). The EPA notes that new evidence since the release of the current, best available peer reviewed scientific assessment for PFNA (ATSDR, 2021) provides further justification for the EPA's analysis of potential economic benefits of PFNA exposure reduction and avoided birth weight effects. More recent epidemiological studies that evaluated PFNA and birth weight, including key studies modeled for PFOA and PFOS (Sagiv et al., 2018; Wikström et al., 2020), as well as a recently published meta-analysis of mean birth weight that indicates the birth weight results for PFNA are robust and consistent, even if associations in some studies may be small in magnitude (Wright et al., 2023). PFNA was modeled in a sensitivity analyses of birth weight benefits. This modeling relied on epidemiological studies published before 2018, representing the current, best available peer reviewed scientific assessment for PFNA (ATSDR, 2021) and the PFAS serum calculator developed by Lu and Bartell (2020) was used to estimate PFNA blood serum levels resulting from PFNA exposures in drinking water.

[e] Liver cancer benefits are not included in the national-level quantified benefits analysis. See appendix O of the EA for the liver cancer benefit analysis results.

The EPA developed PK models to evaluate blood serum PFAS levels in adults resulting from exposure to PFAS via drinking water. To date, the EPA has developed PK models for PFOA and PFOS. The EPA used baseline and regulatory alternative PFOA/PFOS drinking water concentrations as inputs to its PK model to estimate blood serum PFOA/PFOS concentrations for adult males and females. For further detail on the PK model and its application in the EPA's benefits analysis, please see the EPA's *Final Human Health Toxicity Assessments for PFOA and PFOS* (USEPA, 2024c; USEPA, 2024d) and section 6.3 of USEPA (2024g).

1. Quantified Developmental Effects

Exposure to PFOA and PFOS is linked to developmental effects, including decreased infant birth weight (Steenland et al., 2018; Dzierlenga et al., 2020; Verner et al., 2015; USEPA, 2016c; USEPA, 2016d; USEPA, 2024c; USEPA, 2024d; Negri et al., 2017; ATSDR, 2021; Waterfield et al., 2020). The route through which infants are exposed prenatally to PFOA and PFOS is through maternal blood via the placenta. Most studies of the association between maternal serum PFOA/PFOS and birth weight report inverse relationships (Verner et al., 2015; Negri et al., 2017; Steenland et al., 2018; Dzierlenga et al., 2020). The EPA's PK model assumes that mothers were exposed to PFOA/PFOS from birth to the year in which pregnancy occurred.

The EPA quantified and valued changes in birth weight-related risks associated with reductions in exposure to PFOA and PFOS in drinking water. EP-specific time series of the differences between serum PFOA/PFOS concentrations under baseline and regulatory alternatives are inputs into this analysis. For each EP, evaluation of the changes in birth weight impacts involves the following key steps:

1. Estimating the changes in birth weight based on modeled changes in serum PFOA/PFOS levels and exposure-response functions for the effect of serum PFOA/PFOS on birth weight;
2. Estimating the difference in infant mortality probability between the baseline and regulatory alternatives based on changes in birth weight under the regulatory alternatives and the association between birth weight and mortality;
3. Identifying the infant population affected by reduced exposure to PFOA/PFOS in drinking water under the regulatory alternatives;
4. Estimating the changes in the expected number of infant deaths under the regulatory alternatives based on the difference in infant mortality rates and

the population of surviving infants affected by increases in birth weight due to reduced PFOA/PFOS exposure; and

5. Estimating the economic value of reducing infant mortality based on the Value of a Statistical Life and infant morbidity based on reductions in medical costs associated with changes in birth weight for the surviving infants based on the cost of illness.

The EPA also considered the potential benefits from reduced exposure to PFNA that may be realized as a direct result of the final rule. The agency explored the birth weight impacts of PFNA in a sensitivity analysis based on epidemiological studies published before 2018 cited in the current, best available final human health analysis of PFNA (ATSDR, 2021), as well as a recently published meta-analysis of mean birth weight that indicates the birth weight results for PFNA are robust and consistent, even if associations in some studies may be small in magnitude (Wright et al., 2023). The EPA used a unit PFNA reduction scenario (*i.e.,* 1.0 ng/L change) and the PFAS serum calculator developed by Lu and Bartell (2020) to estimate PFNA blood serum levels resulting from PFNA exposures in drinking water. To estimate blood serum PFNA based on its drinking water concentration, the EPA used a first-order single-compartment model whose behavior was previously demonstrated to be consistent with PFOA pharmacokinetics in humans (Bartell et al., 2010). In addition to the PFOA-birth weight and PFOS-birth weight effects analyzed in the EA, the EPA examined the effect of inclusion of PFNA-birth weight effects using estimates from two studies (Lenters et al., 2016; Valvi et al., 2017). The EPA found that inclusion of a 1.0 ng/L PFNA reduction increased annualized birth weight benefits by between a factor of 5.6 to 7.8, relative to the scenario that quantifies a 1.0 ng/L reduction in PFOA and a 1.0 ng/L reduction in PFOS only. The range of estimated PFNA-related increases in benefits is driven by the exposure-response, with smaller estimates produced using the slope factors from Lenters et al. (2016), followed by Valvi et al. (2017). The EPA notes that the PFNA slope factor estimates are orders of magnitude larger than the slope factor estimates used to evaluate the impacts of PFOA/PFOS reductions. The EPA also notes that the PFNA slope factor estimates in this analysis are not precise, with 95 percent CIs covering wide ranges that include zero (*i.e.,* serum PFNA slope factor estimates are not statistically significant at 5 percent level). Caution should be exercised in making judgements about the potential magnitude of change in the national benefits estimates based on the results of these sensitivity analyses, although conclusions about the directionality of these effects can be inferred. The EPA did not include PFNA effects in the national benefits estimates for the final rule because there was insufficient data above the UCMR 3 MRL to reasonably fit model parameters for PFNA. For the EPA's PFNA sensitivity analysis, see appendix K of USEPA (2024g).

To estimate changes in birth weight resulting from reduced exposure to PFOA and PFOS under the regulatory alternatives, the EPA relied on the estimated time series of changes in serum PFOA/PFOS concentrations specific to women of childbearing age and serum-birth weight exposure-response functions provided in recently published meta-analyses. For more detail on the evaluation of the studies used in these meta-analyses, please see the EPA's *Final Human Health Toxicity Assessments for PFOA and PFOS* (USEPA, 2024c; USEPA, 2024d) and section 6.4 of USEPA (2024g).

Changes in serum PFOA and PFOS concentrations are calculated for each PWS EP during each year in the analysis period. The EPA assumes that, given the long half-lives of PFOS and PFOA (with median half-lives of 2.7 and 3.5 years, respectively (Li et al., 2018)), any one-time measurement during or near pregnancy is reflective of a critical exposure window and not subject to considerable error. In other words, blood serum concentrations in a single year are expected to correlate with past exposures and are reflective of maternal exposures regardless of the timing of pregnancy. The mean change in birth weight per increment in long-term PFOA and PFOS exposure is calculated by multiplying each annual change in PFOA and PFOS serum concentration (ng/mL serum) by the PFOA and PFOS serum-birth weight exposure-response slope factors (g birth weight per ng/mL serum) provided in Table 45, respectively. The mean annual change in birth weight attributable to changes in both PFOA and PFOS exposure is the sum of the annual PFOA and PFOS-birth weight change estimates. Additional detail on the derivation of the exposure-response functions can be found in appendix D in USEPA (2024e). appendix K in USEPA (2024e) presents an analysis of birth weight risk reduction considering slope factors specific to the first trimester.

Table 45: Serum Exposure-Birth Weight Response Estimates

Compound	g /ng/mL serum (95% CI)
PFOA [a]	-10.5 (-16.7, -4.4)
PFOS [b]	-3.0 (-4.9, -1.1)

Notes:

[a] The serum-birth weight slope factor for PFOA is based on the main random effects estimate from Steenland et al. (2018).

[b] The serum-birth weight slope factor for PFOS is based on the EPA reanalysis of Dzierlenga et al. (2020).

The EPA places a cap on estimated birth weight changes in excess of 200 g, assuming that such changes in birth weight are unreasonable based on existing studies that found that changes to environmental exposures result in relatively modest birth weight changes (Windham and Fenster, 2008; Klein and Lynch, 2018; Kamai et al., 2019). Modest changes in birth weight even as a result of large changes in PFOA/PFOS serum concentrations may be due to potential bias from studies only including live births (Liew et al., 2015). Additionally, the magnitude of birth weight changes may be correlated with other developmental outcomes such as preterm birth, gestational duration, fetal loss, birth defects, and developmental delays.

Low birth weight is linked to a number of health effects that may be a source of economic burden to society in the form of medical costs, infant mortality, parental and caregiver costs, labor market productivity loss, and education costs (Chaikind and Corman, 1991; Behrman and Butler, 2007; Behrman and Rosenzweig, 2004; Joyce et al., 2012; Kowlessar et al., 2013; Colaizy et al., 2016; Nicoletti et al., 2018; Klein and Lynch, 2018). Recent literature also linked low birth weight to educational attainment and required remediation to improve student outcomes, childhood disability, and future earnings (Jelenkovic et al., 2018; Temple et al., 2010; Elder et al., 2020; Hines et al., 2020; Chatterji et al., 2014; Dobson et al., 2018).

The EPA's analysis focuses on two categories of birth weight impacts that are amenable to monetization associated with incremental changes in birth weight: (1) medical costs associated with changes in infant birth weight and (2) the value of avoiding infant mortality at various birth weights. The birth weight literature related to other sources of economic burden to society (*e.g.,* parental and caregiver costs and productivity losses) is limited in geographic coverage, population size, and range of birth weights evaluated and therefore cannot be used in the EA of birth weight effects from exposure to PFOA/PFOS in drinking water (ICF, 2021).

Two studies showed statistically significant relationships between incremental changes in birth weight and infant mortality: Almond et al. (2005) and Ma and Finch (2010). Ma and Finch (2010) used 2001 NCHS linked birth/infant death data for singleton and multiple birth infants among subpopulations defined by sex and race/ethnicity to estimate a regression model assessing the associations between 14 key birth outcome measures, including birth weight and infant mortality. They found notable variation in the relationship between birth weight and mortality across race/ethnicity subpopulations, with odds ratios for best-fit birth weight-mortality models ranging from 0.8–1 (per 100 g birth weight change). Almond et al. (2005) used 1989–1991 NCHS linked birth/infant death data for multiple birth infants to analyze relationships between birth weight and infant mortality within birth weight increment ranges. For their preferred model, they reported coefficients in deaths per 1,000 births per 1 g increase in birth weight that range from -0.420 to -0.002. However, the data used in these studies (Almond et al., 2005 and Ma and Finch, 2010) are outdated (1989–1991 and 2001, respectively). Given the significant decline in infant mortality over the last 30 years (ICF, 2020) and other maternal and birth characteristics that are likely to influence infant mortality (*e.g.,* average maternal age and rates of maternal smoking), the birth weight-mortality relationship estimates from Almond et al. (2005) and Ma and Finch (2010) are likely to overestimate the benefits of birth weight changes.

Considering the discernible changes in infant mortality over the last 30 years, the EPA developed a regression analysis to estimate the relationship between birth weight and infant mortality using the *Period/Cohort Linked Birth-Infant Death Data Files* published by NCHS from the 2017 period/2016 cohort and the 2018 period/2017 cohort (CDC, 2017; CDC, 2018). These data provide information on infants who are delivered alive and receive a birth certificate. The EPA selected variables of interest for the regression analysis, including maternal demographic and socioeconomic characteristics, maternal risk, and risk mitigation factors (*e.g.,* number of prenatal care visits, smoker status), and infant birth characteristics. The EPA included several variables used in Ma and Finch (2010) (maternal age, maternal education, marital status, and others) as well as additional variables to augment the set of covariates included in the analyses. In addition, the EPA developed separate models for different race/ethnicity categories (non-Hispanic Black, non-Hispanic White, and Hispanic) and interacted birth weight with categories of gestational age, similar to Ma and Finch (2010). Appendix E of USEPA (2024e) provides details on model development and regression results.

Table 46 presents the resulting odds ratios and marginal effects (in terms of deaths per 1,000 births for every 1 g increase in birth weight) estimated for changes in birth weight among different gestational age categories in the mortality regression models for non-Hispanic Black, non-Hispanic White, and Hispanic race/ethnicity subpopulations. Marginal effects for birth weight among gestational age categories vary across different race/ethnicity subpopulations. The marginal effects for birth weight among different gestational age categories are higher in the non-Hispanic Black model than in the non-Hispanic White and Hispanic models, particularly for extremely and very preterm infants, indicating that low birth weight increases the probability of mortality within the first year more so among non-Hispanic Black infants than among non-Hispanic White and Hispanic infants.

The EPA relies on odds ratios estimated using the birth weight-mortality regression model to assess mortality outcomes of reduced exposures to PFOA/PFOS in drinking water under the regulatory alternatives. To obtain odds ratios specific to each race/ethnicity and 100 g birth weight increment considered in the birth weight benefits model,[24] the EPA averaged the estimated odds ratios for 1 g increase in birth weight over the gestational age categories using the number of infants (both singleton and multiple birth) that fall into each gestational age category as weights. Separate gestational age category weights were computed for each 100 g birth weight increment and race/ethnicity subpopulation within the 2017 period/2016 cohort and 2018 period/2017 cohort *Linked Birth-Infant Death Data Files.* The weighted birth weight odds ratios are then used in conjunction with the estimated change in birth weight and baseline infant mortality rates to determine the probability of infant death under the regulatory alternatives, as described further in section 6.4 of USEPA (2024g).

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[24] The birth weight risk reduction model evaluates changes in birth weight in response to PFOA/PFOS drinking water level reductions for infants who fall into 100 g birth weight increments (*e.g.,* birth weight 0–99 g, 100–199 g, 200–299 g. . . 8,000–8,099 g, 8,100–8,165 g).

Table 46: Race/Ethnicity and Gestational Age-Specific Birth Weight Marginal Effects and Odds Ratios from the Mortality Regression Models [1]

Race	Gestational Age Category [2]	Marginal Effect per 1,000 births (95% CI)	Odds Ratio (95% CI)
Non-Hispanic Black	Extremely Preterm	-0.20400 (-0.21910, -0.18890)	0.99817 (0.99802, 0.99832)
	Very Preterm	-0.04580 (-0.04820, -0.04340)	0.99816 (0.99804, 0.99827)
	Moderately Preterm	-0.01030 (-0.01080, -0.009850)	0.99852 (0.99846, 0.99857)
	Term	-0.00453 (-0.00472, -0.00434)	0.99856 (0.99851, 0.9986)
Non-Hispanic White	Extremely Preterm	-0.12160 (-0.13080, -0.11240)	0.99866 (0.99855, 0.99878)
	Very Preterm	-0.03290 (-0.03430, -0.03140)	0.9985 (0.99842, 0.99858)
	Moderately Preterm	-0.00677 (-0.00702, -0.00652)	0.99867 (0.99863, 0.99872)
	Term	-0.00228 (-0.00236, -0.00221)	0.99865 (0.99861, 0.99868)
Hispanic	Extremely Preterm	-0.15260 (-0.16770, -0.13750)	0.99835 (0.99817, 0.99853)
	Very Preterm	-0.03290 (-0.03510, -0.03070)	0.99846 (0.99835, 0.99858)
	Moderately Preterm	-0.00626 (-0.00659, -0.00592)	0.99856 (0.99849, 0.99862)
	Term	-0.00219 (-0.00229, -0.00208)	0.99849 (0.99844, 0.99855)

Notes:

[1] Data based on the *2016/17 and 2017/18 CDC Period Cohort Linked Birth-Infant Death Data Files* obtained from NCHS/National Vital Statistics System (NVSS). Marginal effects and odds ratios are estimated using a regression model that also includes covariates representative of infant birth characteristics in addition to birth weight, maternal demographic characteristics, and maternal risk factors. All effects were statistically significant at the 5 percent level. Additional details are included in appendix E to the EA.

[2] Gestational age categories defined as extremely preterm (<=28 weeks), very preterm (>28 weeks and <=32 weeks), moderately preterm (>32 weeks and <=37 weeks), and term (>37 weeks).

The EPA weighted the race/ethnicity-specific odds ratios in Table 46 by the proportions of the infant populations who fell into each gestational age within a 100 g birth weight increment, based on the 2016/17 and 2017/18 period cohort data, to obtain a weighted odds ratio estimate for each modeled race/ethnicity subpopulation and 100 g birth weight increment.

Based on reduced serum PFOA/PFOS exposures under the regulatory alternatives and the estimated relationship between birth weight and infant mortality, the EPA estimates the subsequent change in birth weight for those infants affected by decreases in PFOA/PFOS and changes in the number of infant deaths. The EPA evaluated these changes at each PWS EP affected by the regulatory alternatives and the calculations are performed for each race/ethnicity group, 100 g birth weight category, and year of the analysis. Additional detail on the calculations the EPA used to estimate changes in birth weight, the affected population size, and infant deaths avoided, and the number of surviving infants is provided in chapter 6 of USEPA (2024g).

The EPA used the Value of a Statistical Life to estimate the benefits of reducing infant mortality and the cost of illness to estimate the economic value of increasing birth weight in the population of surviving infants born to mothers exposed to PFOA and PFOS in drinking water. The EPA's approach to monetizing benefits associated with incremental increases in birth weight resulting from reductions in drinking water PFOA/PFOS levels relies on avoided medical costs associated with various ranges of birth weight. Although the economic burden of treating infants at various birth weights also includes non-medical costs, very few studies to date have quantified such costs (Klein and Lynch, 2018; ICF, 2021). The EPA selected the medical cost function from Klein and Lynch (2018) to monetize benefits associated with the estimated changes in infant birth weight resulting from reduced maternal exposure to PFOA/PFOS.[25]

Using the incremental cost changes from Klein and Lynch (2018), the EPA calculates the change in medical costs resulting from changes in birth weight among infants in the affected population who survived the first year following birth, provided in Table 47.

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[25] The Klein and Lynch (2018) report was externally peer reviewed by three experts with qualifications in economics and public health sciences. The EPA's charge questions to the peer reviewers sought input on the methodology for developing medical cost estimates associated with changes in birth weight. The agency's charge questions, and peer reviewer responses are available in the docket.

Table 47: Simulated Cost Changes for Birth Weight Increases ($2022) (Based on Klein and Lynch, 2018 Table 8)

Birth Weight[a,b]	Simulated Cost Changes for Birth Weight Increases, Dollars per Gram ($2022)[c]		
	+0.04 lb (+18 g)	**+0.11 lb (+50 g)**	**+0.22 lb (+100 g)**
2 lb (907 g)	-$131.66	-$117.44	-$113.82
2.5 lb (1,134 g)	-$98.72	-$88.07	-$85.35
3 lb (1,361 g)	-$74.03	-$66.04	-$64.00
3.3 lb (1,497 g)	-$62.29	-$55.56	-$53.85
4 lb (1,814 g)	-$41.63	-$37.13	-$35.99
4.5 lb (2,041 g)	-$31.21	-$27.84	-$26.98
5 lb (2,268 g)	-$23.41	-$20.88	-$20.23
5.5 lb (2,495 g)	-$0.97	-$0.88	-$0.87
6 lb (2,722 g)	-$0.95	-$0.86	-$0.86
7 lb (3,175 g)	-$0.92	-$0.83	-$0.83
8 lb (3,629 g)	-$0.89	-$0.81	-$0.80
9 lb (4,082 g)	$3.28	$2.99	$3.01
10 lb (4,536 g)	$3.69	$3.37	$3.39

Notes:

[a] Values for birth weight have been converted from lb to g.

[b] Note that simulated medical costs increase, rather than decrease, in response to increased birth weight changes among high birth weight infants (those greater than 8 lb). Among high birth weight infants, there is a higher risk of birth trauma, metabolic issues, and other health problems (Klein and Lynch, 2018).

[c] Values scaled from $2010 to $2022 using the medical care Consumer Price Index (USBLS, 2022).

Tables 48 to 51 provide the health effects avoided and valuation associated with birth weight impacts. The EPA estimated that, over the evaluation period, the final rule will result in annualized benefits from avoided reductions in birth weight of $209 million.

Table 48: National Birth Weight Benefits, Final Rule (PFOA and PFOS MCLs of 4.0 ng/L each, PFHxS, PFNA, and HFPO-DA MCLs of 10 ng/L each, and Hazard Index of 1) (Million $2022)

Benefits Category	**2% Discount Rate**		
	5th Percentile[1]	**Expected Benefits**	**95th Percentile[1]**
Increase in Birth Weight (Millions of Grams)	129.6	216.8	304.1
Number of Birth Weight-Related Deaths Avoided	781.9	1,301.7	1,823.6
Total Annualized Birth Weight Benefits (Million $2022) [2]	$124.85	$209.00	$292.78

Notes:

Detail may not add exactly to total due to independent rounding. Quantifiable benefits are increased under final rule table results relative to the other options presented because of modeled PFHxS occurrence, which results in additional quantified benefits from co-removed PFOA and PFOS.

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.J of this preamble and Table 75. This range does not include the uncertainty described in Table 62.

[2] See Table 72 for a list of the nonquantifiable benefits, and the potential direction of impact these benefits would have on the estimated monetized total annualized benefits in this table.

Table 49: National Birth Weight Benefits, Option 1a (PFOA and PFOS MCLs of 4.0 ng/L) (Million $2022)

Benefits Category	2% Discount Rate		
	5th Percentile[1]	Expected Benefits	95th Percentile[1]
Increase in Birth Weight (Millions of Grams)	128.8	215.6	302.1
Number of Birth Weight-Related Deaths Avoided	777.4	1,294.4	1,812.9
Total Annualized Birth Weight Benefits (Million $2022) [2]	$124.82	$207.82	$291.00

Notes:

Detail may not add exactly to total due to independent rounding.

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.J of this preamble and Table 75. This range does not include the uncertainty described in Table 62.

[2] See Table 72 for a list of the nonquantifiable benefits, and the potential direction of impact these benefits would have on the estimated monetized total annualized benefits in this table.

Table 50: National Birth Weight Benefits, Option 1b (PFOA and PFOS MCLs of 5.0 ng/L) (Million $2022)

Benefits Category	2% Discount Rate		
	5th Percentile[1]	Expected Benefits	95th Percentile[1]
Increase in Birth Weight (Millions of Grams)	111.3	185.6	260.3
Number of Birth Weight-Related Deaths Avoided	668.9	1,114.7	1,561.2
Total Annualized Birth Weight Benefits (Million $2022) [2]	$107.34	$178.97	$250.00

Notes:

Detail may not add exactly to total due to independent rounding.

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.J of this preamble and Table 75. This range does not include the uncertainty described in Table 62.

[2] See Table 72 for a list of the nonquantifiable benefits, and the potential direction of impact these benefits would have on the estimated monetized total annualized benefits in this table.

Table 51: National Birth Weight Benefits, Option 1c (PFOA and PFOS MCLs of 10.0 ng/L) (Million $2022)

Benefits Category	2% Discount Rate		
	5th Percentile[1]	Expected Benefits	95th Percentile[1]
Increase in Birth Weight (Millions of Grams)	62.1	102.0	142.4
Number of Birth Weight-Related Deaths Avoided	375.8	616.6	859.1
Total Annualized Birth Weight Benefits (Million $2022) [2]	$60.24	$98.97	$137.75

Notes:

Detail may not add exactly to total due to independent rounding.

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.J of this preamble and Table 75. This range does not include the uncertainty described in Table 62.

[2] See Table 72 for a list of the nonquantifiable benefits, and the potential direction of impact these benefits would have on the estimated monetized total annualized benefits in this table.

2. Quantified Cardiovascular Effects

CVD is one of the leading causes of premature mortality in the United States (D'Agostino et al., 2008; Goff et al., 2014; Lloyd-Jones et al., 2017). As discussed in the EPA's *Final Human Health Toxicity Assessments for PFOA and PFOS,* exposure to PFOA and PFOS through drinking water contributes to increased serum PFOA and PFOS concentrations and elevated levels of TC, as well as suggestive evidence of changes in levels of HDLC and elevated levels of systolic blood pressure (USEPA, 2024c; USEPA, 2024d). Changes in TC and blood pressure are associated with changes in incidence of CVD events such as myocardial infarction (*i.e.,* heart attack), ischemic stroke, and cardiovascular mortality occurring in populations without prior CVD event experience (D'Agostino et al., 2008; Goff et al., 2014; Lloyd-Jones et al., 2017).

The EPA recognizes that the epidemiologic literature that provides strong support for an effect of PFOA and PFOS on cholesterol and blood pressure does not provide direct support for an effect of PFOA and PFOS on the risk of CVD. Therefore, the EPA uses the approach outlined here to link changes in CVD risk biomarkers (*i.e.,* cholesterol and blood pressure) to changes in CVD risk.

For each EP, evaluation of the changes in CVD risk involves the following key steps:

1. Estimation of annual changes in TC and blood pressure levels using exposure-response functions for the potential effects of serum PFOA/PFOS on these biomarkers;
2. Estimation of the annual incidence of fatal and non-fatal first hard CVD events, defined as fatal and non-fatal myocardial infarction, fatal and non-fatal ischemic stroke or other coronary heart disease death occurring in populations without prior CVD event experience (D'Agostino et al., 2008; Goff et al., 2014; Lloyd-Jones et al., 2017), and post-acute CVD mortality corresponding to baseline and regulatory alternative TC and blood pressure levels in all populations alive during or born after the start of the evaluation period; and
3. Estimation of the economic value of reducing CVD mortality and morbidity from baseline to regulatory alternative levels, using the Value of a Statistical Life and cost of illness measures, respectively.

Given the breadth of evidence linking PFOA and PFOS exposure to effects on TC and blood pressure in general adult populations, the EPA quantified public health impacts of changes in these well-established CVD risk biomarkers (D'Agostino et al., 2008; Goff et al., 2014; Lloyd-Jones et al., 2017) by estimating changes in incidence of several CVD events. Specifically, the EPA assumed that PFOA/PFOS-related changes in TC and blood pressure had the same effect on the CVD risk as the changes unrelated to chemical exposure and used the Pooled Cohort ASCVD model (Goff et al., 2014) to evaluate their impacts on the incidence of myocardial infarction, ischemic stroke, and cardiovascular mortality occurring in populations without prior CVD event experience.

The ASCVD model includes TC as a predictor of first hard CVD events. The EPA did not identify any readily available relationships for PFOA or PFOS and TC that were specifically relevant to the age group of interest (40–89 years, the years for which the ASCVD model estimates the probability of a first hard CVD event). Therefore, the agency developed a meta-analysis of studies reporting associations between serum PFOA or PFOS and TC in general populations (*e.g.,* populations that are not a subset of workers or pregnant women). Statistical analyses that combine the results of multiple studies, such as meta-analyses, are widely applied to investigate the associations between contaminant levels and associated health effects. Such analyses are suitable for economic assessments because they can improve precision and statistical power (Engels et al., 2000; Deeks, 2002; Rücker et al., 2009).

The EPA identified 14 studies from which to derive slope estimates for PFOA and PFOS associations with serum TC levels. Appendix F of USEPA (2024e) provides further detail on the studies selection criteria, meta-data development, meta-analysis results, and discussion of the uncertainty and limitations inherent in the EPA's exposure-response analysis.

The EPA developed exposure-response relationships between serum PFOA/PFOS and TC for use in the CVD analysis using the meta-analyses restricted to studies of adults in the general population reporting similar models. When using studies reporting linear associations between TC and serum PFOA or PFOS, the EPA estimated a positive increase in TC of 1.57 (95 percent CI: 0.02, 3.13) mg/dL per ng/mL serum PFOA (p-value=0.048), and of 0.08 (95 percent CI: -0.01, 0.16) mg/dL per ng/mL serum PFOS (p-value=0.064). Based on the systematic review conducted by the EPA to develop the EPA's *Final Human Health Toxicity Assessments for PFOA and PFOS,* the available evidence supports a positive association between PFOS and TC in the general population. For more information on the systematic review and results, see USEPA (2024c) and USEPA (2024d).

PFOS exposure has been linked to other cardiovascular outcomes, such as systolic blood pressure and hypertension (Liao et al., 2020; USEPA, 2024d). Because systolic blood pressure is another predictor used by the ASCVD model, the EPA included the estimated changes in blood pressure from reduced exposure to PFOS in the CVD analysis. The EPA selected the slope from the Liao et al. (2020) study—a high confidence study conducted based on U.S. general population data from NHANES cycles 2003–2012. The evidence on the associations between PFOA and blood pressure is not as consistent as for PFOS. Therefore, the EPA is not including effect estimates for the serum PFOA-blood pressure associations in the CVD analysis.

The EPA relies on the life table-based approach to estimate CVD risk reductions because (1) changes in serum PFOA/PFOS in response to changes in drinking water PFOA/PFOS occur over multiple years, (2) CVD risk, relying on the ASCVD model, can be modeled only for those older than 40 years without prior CVD history, and (3) individuals who have experienced non-fatal CVD events have elevated mortality implications immediately and within at least five years of the first occurrence. Recurrent life table calculations are used to estimate a PWS EP-specific annual time series of CVD event incidence for a population cohort characterized by sex, race/ethnicity, birth year, age at the start of the PFOA/PFOS evaluation period (*i.e.,* 2024), and age- and sex-specific time series of changes in TC and blood pressure levels obtained by combining serum PFOA/PFOS concentration time series with exposure-response information. Baseline and regulatory alternatives are evaluated separately, with regulatory alternative TC and blood pressure levels estimated using baseline information on these biomarkers from external statistical data sources and modeled changes in TC and blood pressure due to conditions under the regulatory alternatives.

The EPA estimated the incidence of first hard CVD events based on TC serum and blood pressure levels using the ASCVD model (Goff et al., 2014), which predicts the 10-year probability of a hard CVD event to be experienced by a person without a prior CVD history. The EPA adjusted the modeled population cohort to exclude individuals with pre-existing conditions, as the ASCVD risk model does not apply to these individuals. For blood pressure effects estimation, the EPA further restricts the modeled population to those not using antihypertensive medications for consistency with the exposure-response relationship. Modeled first hard CVD events include fatal and non-fatal myocardial infarction, fatal and non-fatal ischemic stroke, and other coronary heart disease mortality. The EPA has also estimated the incidence of post-acute CVD mortality among survivors of the first myocardial infarction or ischemic stroke within 6 years of the initial event.

The estimated CVD risk reduction resulting from reducing serum PFOA and serum PFOS concentrations is the difference in annual incidence of CVD events (*i.e.,* mortality and morbidity associated with first-time CVD events and post-acute CVD mortality) under the baseline and regulatory alternatives. Appendix G of USEPA (2024e) provides detailed information on all CVD model components, computations, and sources of data used in modeling.

The EPA uses the Value of a Statistical Life to estimate the benefits of reducing mortality associated with hard CVD events in the population exposed to PFOA and PFOS in drinking water. The EPA relies on cost of illness-based valuation that represents the medical costs of treating or mitigating non-fatal first hard CVD events (myocardial infarction, ischemic stroke) during the three years following an event among those without prior CVD history, adjusted for post-acute mortality.

The annual medical expenditure estimates for myocardial infarction and ischemic stroke are based on O'Sullivan et al. (2011). The estimated expenditures do not include long-term institutional and home health care. For non-fatal myocardial infarction, O'Sullivan et al. (2011) estimated medical expenditures are $53,246 ($2022) for the initial event and then $33,162, $14,635, $13,078 annually within 1, 2, and 3 years after the initial event, respectively. For non-fatal ischemic stroke, O'Sullivan et al. (2011) estimated medical expenditures are $16,503 ($2022) for the initial event and then $11,988, $788, $1,868 annually within 1, 2, and 3 years after the initial event, respectively. Annual estimates within 1, 2, and 3 years after the initial event include the incidence of secondary CVD events among survivors of first myocardial infarction and ischemic stroke events.

To estimate the present discounted value of medical expenditures within 3 years of the initial non-fatal myocardial infarction, the EPA combined O'Sullivan et al. (2011) myocardial infarction-specific estimates with post-acute survival probabilities based on Thom et al. (2001) (for myocardial infarction survivors aged 40–64) and Li et al. (2019) (for myocardial infarction survivors aged 65+). To estimate the present discounted value of medical expenditures within 3 years of the initial non-fatal ischemic stroke, the EPA combined O'Sullivan et al. (2011) ischemic stroke-specific estimates with post-acute survival probabilities based on Thom et al. (2001) (for ischemic stroke survivors aged 40–64, assuming post-acute myocardial infarction survival probabilities reasonably approximate post-acute ischemic stroke survival probabilities) and Li et al. (2019) (for ischemic stroke survivors aged 65+). The EPA did not identify post-acute ischemic stroke mortality information in this age group, but instead applied post-acute myocardial infarction mortality estimates for ischemic stroke valuation. Table 52 presents the resulting myocardial infarction and ischemic stroke unit values.

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Table 52: Cost of Illness of Non-Fatal First CVD Event Used in Modeling

Type of First Non-fatal Hard CVD Event	Age Group	Present Discounted Value of 3-Year Medical Expenditures ($2022, 2% discount rate)[a,b] Adjusted for Post-Acute Mortality[c]
MI	40-64 years	$110,040
	65+ years	$96,626
IS	40-64 years	$30,373
	65+ years	$27,954

Abbreviations: CVD – cardiovascular disease; MI – myocardial infarction (ICD9=410; ICD10=I21), IS – ischemic stroke (ICD9=433, 434; ICD10=I63).

Notes:

[a] Estimates of annual medical expenditures are from O'Sullivan et al. (2011).

[b] Original values from O'Sullivan et al. (2011) were inflated to $2022 using the medical care Consumer Price Index (USBLS, 2022).

[c] Post-acute MI mortality data for those aged 40-64 years is from Thom et al. (2001); probabilities to survive 1 year, 2 years, and 3 years after the initial event are 0.93, 0.92, and 0.90, respectively. The EPA applies these mortality values to derive the IS value in this age group. Post-acute MI mortality data and post-acute IS mortality data for persons aged 65 years and older are from Li et al. (2019). For MI, probabilities to survive 1 year, 2 years, and 3 years after the initial event are 0.68, 0.57, and 0.49, respectively. For IS, probabilities to survive 1 year, 2 years, and 3 years after the initial event are 0.67, 0.57, and 0.48, respectively.

Tables 53 to 56 provide the health effects avoided and valuation associated with CVD. The EPA estimated that, over the evaluation period, the final rule will result in annualized benefits from avoided CVD cases and deaths of $606 million.

Table 53: National CVD Benefits, Final Rule (PFOA and PFOS MCLs of 4.0 ng/L each, PFHxS, PFNA, and HFPO-DA MCLs of 10 ng/L each, and Hazard Index of 1) (Million $2022)

Benefits Category	**2% Discount Rate**		
	5th Percentile[1]	**Expected Benefits**	**95th Percentile[1]**
Number of Non-Fatal MI Cases Avoided	1,407.7	6,333.1	11,189.0
Number of Non-Fatal IS Cases Avoided	2,074.8	9,247.6	16,279.0
Number of CVD Deaths Avoided	845.5	3,715.8	6,555.6
Total Annualized CVD Benefits (Million $2022) [2]	$140.66	$606.09	$1,069.40

Notes:

Detail may not add exactly to total due to independent rounding. Quantifiable benefits are increased under final rule table results relative to the other options presented because of modeled PFHxS occurrence, which results in additional quantified benefits from co-removed PFOA and PFOS.

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.J of this preamble and Table 75. This range does not include the uncertainty described in Table 62.

[2] See Table 72 for a list of the nonquantifiable benefits, and the potential direction of impact these benefits would have on the estimated monetized total annualized benefits in this table.

Table 54: National CVD Benefits, Option 1a (PFOA and PFOS MCLs of 4.0 ng/L)

(Million $2022)

Benefits Category	2% Discount Rate		
	5th Percentile[1]	Expected Benefits	95th Percentile[1]
Number of Non-Fatal MI Cases Avoided	1,400.8	6,296.0	11,115.0
Number of Non-Fatal IS Cases Avoided	2,065.0	9,194.8	16,203.0
Number of CVD Deaths Avoided	839.9	3,695.1	6,484.4
Total Annualized CVD Benefits (Million $2022) [2]	$140.12	$602.72	$1,059.60

Notes:

Detail may not add exactly to total due to independent rounding.

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.J of this preamble and Table 75. This range does not include the uncertainty described in Table 62.

[2] See Table 72 for a list of the nonquantifiable benefits, and the potential direction of impact these benefits would have on the estimated monetized total annualized benefits in this table.

Table 55: National CVD Benefits, Option 1b (PFOA and PFOS MCLs of 5.0 ng/L)

(Million $2022)

Benefits Category	2% Discount Rate		
	5th Percentile[1]	Expected Benefits	95th Percentile[1]
Number of Non-Fatal MI Cases Avoided	1,209.2	5,352.0	9,417.5
Number of Non-Fatal IS Cases Avoided	1,778.3	7,826.9	13,778.0
Number of CVD Deaths Avoided	733.1	3,146.8	5,518.0
Total Annualized CVD Benefits (Million $2022) [2]	$119.18	$513.27	$900.13

Notes:

Detail may not add exactly to total due to independent rounding.

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.J of this preamble and Table 75. This range does not include the uncertainty described in Table 62.

[2] See Table 72 for a list of the nonquantifiable benefits, and the potential direction of impact these benefits would have on the estimated monetized total annualized benefits in this table.

Table 56: National CVD Benefits, Option 1c (PFOA and PFOS MCLs of 10.0 ng/L)

(Million $2022)

Benefits Category	**2% Discount Rate**		
	5th Percentile[1]	**Expected Benefits**	**95th Percentile**[1]
Number of Non-Fatal MI Cases Avoided	673.7	2,776.5	4,872.8
Number of Non-Fatal IS Cases Avoided	987.0	4,079.2	7,145.6
Number of CVD Deaths Avoided	411.6	1,640.9	2,878.1
Total Annualized CVD Benefits (Million $2022) [2]	$66.97	$267.56	$469.05

Notes:

Detail may not add exactly to total due to independent rounding.

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.J of this preamble and Table 75. This range does not include the uncertainty described in Table 62.

[2] See Table 72 for a list of the nonquantifiable benefits, and the potential direction of impact these benefits would have on the estimated monetized total annualized benefits in this table.

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3. Quantified Kidney Cancer Effects

Data on the association between PFOA exposure and kidney cancer (*i.e.,* RCC), particularly from epidemiological studies, indicate a positive association between exposure and increased risk of RCC. Epidemiology studies indicated that exposure to PFOA was associated with an increased risk of RCC (CalEPA, 2021; ATSDR, 2021; USEPA, 2016c; USEPA, 2024c, USEPA, 2024j). In the PFOA HESD (USEPA, 2016c), the EPA determined that PFOA is likely to be carcinogenic to humans (USEPA, 2005a) based in part on evidence of associations between PFOA exposure and kidney cancer in humans. A recent study of the relationship between PFOA and RCC in U.S. general populations found strong evidence of a positive association between exposure to PFOA and RCC in humans (Shearer et al., 2021). A meta-analysis of epidemiological literature also concluded that there was an increased risk of kidney cancer associated with increased PFOA serum concentrations (Bartell and Vieira, 2021). As such, the EPA selected RCC as a key outcome when assessing the health impacts of reduced PFOA exposures.

The EPA quantified and valued the changes in RCC risk associated with reductions in serum PFOA levels that are in turn associated with reductions in drinking water PFOA concentrations under the regulatory alternatives. PWS EP-specific time series of the differences between serum PFOA concentrations under baseline and regulatory alternatives are inputs into this analysis. For each PWS EP, evaluation of the changes in RCC impacts involves the following key steps:

1. Estimating the changes in RCC risk based on modeled changes in serum PFOA levels and the exposure-response function for the effect of serum PFOA on RCC;
2. Estimating the annual incidence of RCC cases and excess mortality among those with RCC in all populations corresponding to baseline and regulatory alternative RCC risk levels, as well as estimating the regulatory alternative-specific reduction in cases relative to the baseline, and
3. Estimating the economic value of reducing RCC mortality from baseline to regulatory alternative levels, using the Value of a Statistical Life and cost of illness measures, respectively.

To identify an exposure-response function, the EPA reviewed studies highlighted in the HESD for PFOA (USEPA, 2016c) and a recent study discussed in both the California Environmental Protection Agency's Office of Environmental Health Hazard Assessment (OEHHA) *PFOA Public Health Goals* report (CalEPA, 2021) and the EPA's *Final Human Health Toxicity Assessment for PFOA* (USEPA, 2024c; USEPA, 2024j). Steenland et al. (2015) observed an increase in kidney cancer deaths among workers with high exposures to PFOA. Vieira et al. (2013) found that kidney cancer was positively associated with ''high'' and ''very high'' PFOA exposures. Barry et al. (2013) found a slight trend in cumulative PFOA serum exposures and kidney cancer among the C8 Health Project population. In a large case-control general population study of the relationship between PFOA and kidney cancer in 10 locations across the U.S., Shearer et al. (2021) found evidence that exposure to PFOA is associated with RCC, the most common form of kidney cancer, in humans.

To evaluate changes between baseline and regulatory alternative RCC risk resulting from reduced exposure to PFOA, the EPA relied on the estimated time series of changes in serum PFOA concentrations (section 6.3) and the serum-RCC exposure-response function provided by Shearer et al. (2021): 0.00178 $(ng/mL)^{-1}$. The analysis reported in Shearer et al. (2021) was designed as a case-control study with population controls based on 10 sites within the U.S. population. Shearer et al. (2021) accounted for age, sex, race, ethnicity, study center, year of blood draw, smoking, and hypertension in modeling the association between PFOA and RCC. Results showed a strong and statistically significant association between PFOA and RCC. The EPA selected the exposure-response relationship from Shearer et al. (2021) because it included exposure levels typical in the general population and the study was found to have a low risk of bias when assessed in the EPA's *Final Human Health Toxicity Assessment for PFOA* (USEPA, 2024c; USEPA, 2024j).

The linear slope factor developed by the agency (see section 4.2 of USEPA, 2024c) based on Shearer et al. (2021) enables estimation of the changes in the lifetime RCC risk associated with reduced lifetime serum PFOA levels. Because baseline RCC incidence statistics are not readily available from the National Cancer Institute (NCI) public use data, the EPA used kidney cancer statistics in conjunction with an assumption that RCC comprises 90 percent of all kidney cancer cases to estimate baseline lifetime probability of RCC (USEPA, 2024c; American Cancer Society, 2020). The EPA estimated the baseline lifetime RCC incidence for males at 1.89 percent and the baseline lifetime RCC incidence for females at 1.05 percent. Details of these calculations are provided in appendix H of USEPA (2024e).

Similar to the EPA's approach for estimating CVD risk reductions, the EPA relies on the life table approach to estimate RCC risk reductions. The outputs of the life table calculations are the PWS EP-specific estimates of the annual change in the number of RCC cases and the annual change in excess RCC population mortality. For more detail on the EPA's application of the life table to cancer benefits analyses, please see appendix H of USEPA (2024e).

Although the change in PFOA exposure likely affects the risk of developing RCC beyond the end of the analysis period (the majority of RCC cases manifest during the latter half of the average individual lifespan; see appendix H of USEPA (2024e), the EPA does not capture effects after the end of the period of analysis, 2105. Individuals alive after the end of the period of analysis likely benefit from lower lifetime exposure to PFOA. Lifetime health risk model data sources include the EPA SDWIS, age-, sex-, and race/ethnicity-specific population estimates from the U.S. Census Bureau (2020), the Surveillance, Epidemiology, and End Results (SEER) program database (Surveillance Research Program—National Cancer Institute, 2020a; National Cancer Institute, 2020b), and the CDC NCHS. Appendix H of USEPA (2024e) provides additional detail on the data sources and information used in this analysis as well as baseline kidney cancer statistics. Appendix B of USEPA (2024e) describes estimation of the affected population.

The EPA uses the Value of a Statistical Life to estimate the benefits of reducing mortality associated with RCC in the population exposed to PFOA in drinking water. The EPA uses the cost of illness-based valuation to estimate the benefits of reducing morbidity associated with RCC.

The EPA used the medical cost information from a recent RCC cost-effectiveness study by Ambavane et al. (2020) to develop cost of illness estimates for RCC morbidity. Ambavane et al. (2020) used a discrete event simulation model to estimate the lifetime treatment costs of several RCC treatment sequences, which included first and second line treatment medication costs, medication administration costs, adverse effect management costs, and disease management costs on- and off-treatment. To this end, the authors combined RCC cohort data from CheckMate 214 clinical trial and recent US-based healthcare cost information assembled from multiple sources (see supplementary information from Ambavane et al. (2020)).

The EPA received public comments on the EA for the proposed rule related to the EPA's use of cost of illness information for morbidity valuation. Specifically, some commenters recommended that the EPA use willingness to pay information (instead of cost of illness information) when valuing the costs associated with non-fatal illnesses, stating that willingness to pay information better accounts for lost opportunity costs (*e.g.,* lost productivity and pain and suffering) associated with non-fatal illnesses (USEPA, 2024k). To better account for these opportunity costs, the EPA used recently available willingness to pay values in a sensitivity analysis for morbidity associated with RCC. The sensitivity analysis results show that when willingness to pay values are used in RCC benefits analysis, morbidity benefits are increased by approximately 2 percent. See appendix O of the EA for full details and results on the willingness to pay sensitivity analyses.

Table 57 summarizes RCC morbidity cost of illness estimates derived by the EPA using Ambavane et al. (2020)-reported disease management costs on- and off-treatment along with medication, administration, and adverse effect management costs for the first line treatment that initiated the most cost-effective treatment sequences as identified by Ambavane et al. (2020), *i.e.,* the nivolumab and ipilimumab drug combination. This is a forward-looking valuation approach in that it assumes that the clinical practice would follow the treatment recommendations in Ambavane et al. (2020) and other recent studies cited therein. The EPA notes that the second line treatment costs are not reflected in the EPA's cost of illness estimates, because Ambavane et al. (2020) did not report information on the expected durations of the treatment-free interval (between the first line treatment discontinuation and the second line treatment initiation) and the second line treatment phase, conditional on survival beyond discontinuation of the second line treatment. As such, the EPA valued RCC morbidity at $261,175 ($2022) during year 1 of the diagnosis, $198,705 ($2022) during year 2 of the diagnosis, and $1,661 ($2022) starting from year 3 of the diagnosis. Additionally, the EPA assumed that for individuals with RCC who die during the specific year, the entire year-specific cancer treatment regimen is applied prior to the death event. This may overestimate benefits if a person does not survive the entire year.

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Table 57: RCC Morbidity Valuation

Time Interval	First Line Medication ($2018)[a]	First Line Administration ($2018)[a]	First Line Adverse Effect Management	Disease Management ($2018)[a]	Total ($2018)	Total ($2022)[d]
Monthly cost, month 1-3 from diagnosis[a,e]	32,485	516	78	73	33,152	37,382
Monthly cost, month 4-24 from diagnosis[b,f]	13,887	647	78	73	14,685	16,559
Monthly cost, month 25+ from diagnosis[g]	-	-	-	123	123	139
Annual cost, year 1 from diagnosis	222,438	7,371	934	878	231,621	261,175
Annual cost, year 2 from diagnosis	166,644	7,764	934	878	176,220	198,705
Annual cost, year 3+ from diagnosis	-	-	-	1,473	1,473	1,661

Notes:

[a] Ambavane et al. (2020) Table 1.

[b] Ambavane et al. (2020) p. 41, a maximum treatment duration assumption of 2 years.

[c] The adverse effect management costs of $1,868 in Ambavane et al. (2020) Table 1 were reported for the treatment duration. The EPA used the treatment duration of 24 months (i.e., 2 years) to derive monthly costs of $77.83.

[d] To adjust for inflation, the EPA used U.S. Bureau of Labor Statistics Consumer Price Index for All Urban Consumers: Medical Care Services in U.S. (City Average).

[e] First line treatment induction.

[f] First line treatment maintenance.

[g] Treatment-free interval.

Tables 58 to 61 provide the health effects avoided and valuation associated with RCC. The EPA estimated that, over the evaluation period, the final rule will result in annualized benefits from avoided RCC cases and deaths of $354 million.

Table 58: National RCC Benefits, Final Rule (PFOA and PFOS MCLs of 4.0 ng/L each, PFHxS, PFNA, and HFPO-DA MCLs of 10 ng/L each, and Hazard Index of 1) (Million $2022)

Benefits Category	**2% Discount Rate**		
	5th Percentile[1]	**Expected Benefits**	**95th Percentile[1]**
Number of Non-Fatal RCC Cases Avoided	1,091.5	6,964.2	17,937.0
Number of RCC-Related Deaths Avoided	320.4	2,028.8	5,206.5
Total Annualized RCC Benefits (Million $2022) [2, 3]	$61.33	$353.90	$883.55

Notes:

Detail may not add exactly to total due to independent rounding. Quantifiable benefits are increased under final rule table results relative to the other options presented because of modeled PFHxS occurrence, which results in additional quantified benefits from co-removed PFOA and PFOS.

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.J of this preamble and Table 75. This range does not include the uncertainty described in Table 62.

[2] See Table 72 for a list of the nonquantifiable benefits, and the potential direction of impact these benefits would have on the estimated monetized total annualized benefits in this table.

[3] When using willingness-to-pay metrics to monetize morbidity benefits, total annualized RCC benefits are increased by $7.1 million.

Table 59: National RCC Benefits, Option 1a (PFOA and PFOS MCLs of 4.0 ng/L)

(Million $2022)

Benefits Category	**2% Discount Rate**		
	5th Percentile[1]	**Expected Benefits**	**95th Percentile[1]**
Number of Non-Fatal RCC Cases Avoided	1,082.0	6,922.4	17,870.0
Number of RCC-Related Deaths Avoided	319.1	2,016.7	5,190.9
Total Annualized RCC Benefits (Million $2022) [2]	$60.90	$351.79	$877.47

Notes: Detail may not add exactly to total due to independent rounding.

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.J of this preamble and Table 75. This range does not include the uncertainty described in Table 62.

[2] See Table 72 for a list of the nonquantifiable benefits, and the potential direction of impact these benefits would have on the estimated monetized total annualized benefits in this table.

Table 60: National RCC Benefits, Option 1b (PFOA and PFOS MCLs of 5.0 ng/L)

(Million $2022)

Benefits Category	**2% Discount Rate**		
	5th Percentile[1]	**Expected Benefits**	**95th Percentile[1]**
Number of Non-Fatal RCC Cases Avoided	851.9	5,696.1	14,906.0
Number of RCC-Related Deaths Avoided	251.6	1,663.8	4,328.4
Total Annualized RCC Benefits (Million $2022) [2]	$48.41	$290.72	$730.99

Notes:

Detail may not add exactly to total due to independent rounding.

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.J of this preamble and Table 75. This range does not include the uncertainty described in Table 62.

[2] See Table 72 for a list of the nonquantifiable benefits, and the potential direction of impact these benefits would have on the estimated monetized total annualized benefits in this table.

Table 61: National RCC Benefits, Option 1c (PFOA and PFOS MCLs of 10.0 ng/L)

(Million $2022)

Benefits Category	**2% Discount Rate**		
	5th Percentile[1]	**Expected Benefits**	**95th Percentile**[1]
Number of Non-Fatal RCC Cases Avoided	372.1	2,648.1	6,967.4
Number of RCC-Related Deaths Avoided	111.5	782.8	2,057.3
Total Annualized RCC Benefits (Million $2022) [2]	$21.20	$137.30	$352.07

Notes:

Detail may not add exactly to total due to independent rounding.

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.J of this preamble and Table 75. This range does not include the uncertainty described in Table 62.

[2] See Table 72 for a list of the nonquantifiable benefits, and the potential direction of impact these benefits would have on the estimated monetized total annualized benefits in this table.

4. Key Limitations and Uncertainties in the Benefits Analysis

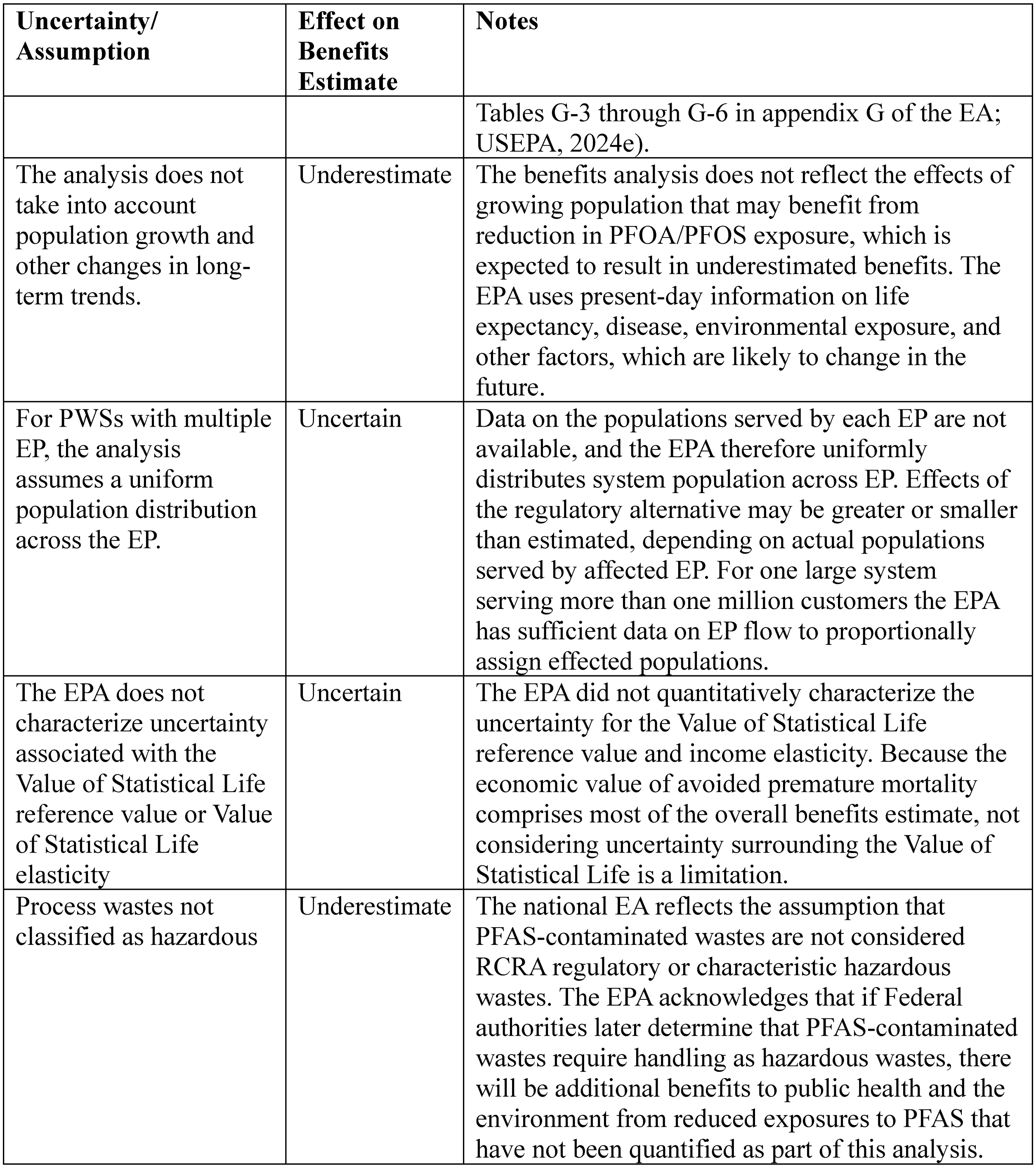
The following section discusses the uncertainty information incorporated in the quantitative benefits analysis. There are additional sources of uncertainty and limitations that could not be modeled quantitatively as part of the national benefits analysis. These sources of uncertainty are characterized in detail in section 6.8 of USEPA (2024g). This summary includes uncertainties that are specific to application of PK models for blood serum PFAS concentration estimation, developmental effects (*i.e.,* infant birth weight) modeling, CVD impacts modeling, RCC impacts modeling, and modeling of bladder cancer impacts from GAC treatment-related reductions in the sum of four trihalomethanes (THM4). Table 62 presents the key limitations and uncertainties that apply to the benefits analysis for the final rule. The EPA notes that in most cases it is not possible to judge the extent to which a particular limitation or uncertainty could affect the magnitude of the estimated benefits. Therefore, in each of the following tables, the EPA notes the potential direction of the impact on the quantified benefits (*e.g.,* a source of uncertainty that tends to underestimate quantified benefits indicates expectation for larger quantified benefits) but does not prioritize the entries with respect to the impact magnitude.

Table 62: Key Limitations and Uncertainties that Apply to Benefits Analyses

Considered for the Final PFAS Rule

Uncertainty/ Assumption	Effect on Benefits Estimate	Notes
The EPA has quantified benefits for three health endpoints for PFOA (birth weight, CVD, and RCC) and two health endpoints for PFOS (birth weight and CVD)	Underestimate	For various reasons, the EPA has not quantified the benefit of removing PFOA and PFOS from drinking water for most of the health endpoints PFOA and PFOS are expected to impact. See discussion in section F for more information about these nonquantifiable benefits.
The EPA has only quantified benefits for one co-removed contaminant group (THM4)	Underestimate	Treatment technologies that remove PFAS can also remove numerous other contaminants, including some other PFAS compounds, additional regulated and unregulated DBPs, heavy metals, organic contaminants, pesticides, among others. These co-removal benefits may be significant, depending on co-occurrence, how many facilities install treatment and which treatment option they select.
The EPA has not quantified national benefits for any health endpoint for the PFAS that make up the Hazard Index (PFHxS, PFNA, HFPO-DA, and PFBS)	Underestimate	PFHxS, PFNA, HFPO-DA, and PFBS each have substantial health impacts on multiple health endpoints. However, the effects of PFNA on birth weight are evaluated as part of a sensitivity analysis in appendix K. See discussion in section D for more information about these nonquantifiable benefits.
The analysis considers PFOA/PFOS concentrations from NTNCWSs	Overestimate	SDWIS population served estimates for NTNCWSs represent both the population that has regular exposure to the NTNCWS' drinking water (e.g., the employees at a location) and the peak day transient population (e.g., customers) who have infrequent exposure to the NTNCWS' drinking water. Estimating the demographic distribution and the share of daily drinking water consumption for these two types of NTNCWS populations would be difficult across many of the industries which operate NTNCWSs. The inclusion of NTNCWS results is an overestimate of benefits because daily drinking water consumption for these populations is also modeled at their residential CWS.
The EPA assumes that the effects of PFOA and	Uncertain	The exposure-response functions used in benefits analyses assume that the effects of serum PFOA/PFOS on the health outcomes considered

Uncertainty/ Assumption	**Effect on Benefits Estimate**	**Notes**
PFOS exposures are independent.		are independent and therefore additive. This assumption is consistent with the *Framework for Estimating Noncancer Health Risks Associated with Mixtures of Per- and Polyfluoroalkyl Substances (PFAS) (*USEPA, 2024a*).* Due to limited evidence, the EPA does not consider synergies or antagonisms in PFOA/PFOS exposure-response.
The derivation of PFOA/PFOS exposure-response functions for the relationship between PFOA/PFOS serum and associated health outcomes assumes that there are no threshold serum concentrations below which effects do not occur.	Overestimate	The new data and the EPA's Final Human Health Toxicity Assessments indicate that the levels at which adverse health effects could occur are much lower than previously understood when the EPA issued the 2016 health advisories for PFOA and PFOS (70 ng/L) – including near zero for certain health effects. Therefore, the exposure-response functions used in benefits analyses assume that there are no threshold serum concentrations below which effects do not occur. This could result in a slight overestimate of benefits for noncancer health endpoints.
Causality is assumed for all health effects for which exposure-response functions are used to estimate risk.	Overestimate	Analyses evaluating the evidence on the associations between PFAS exposure and health outcomes are ongoing and the EPA has not conclusively determined causality. As described in section 6.2 of the EA, the EPA modeled health risks from PFOA/PFOS exposure for endpoints for which the evidence of association was found to be likely. These endpoints include birth weight, TC, and RCC. While the evidence supporting causality between DBP exposure and bladder cancer has increased since the EPA's Stage 2 DBP Rule (NTP, 2021; Weisman et al., 2022), causality has not yet been conclusively determined (Regli et al., 2015).
The analysis assumes that quantified benefits categories are additive.	Uncertain	The EPA did not model birth weight, CVD, RCC, and bladder cancer benefits jointly, in a competing risk framework. Therefore, reductions in health risk in a specific benefits category do not influence health risk reductions in another benefits category. For example, lower risk of CVD and associated mortality implies a larger population that could benefit from cancer risk reductions, because cancer incidence grows considerably later in life (see



Uncertainty/ Assumption	Effect on Benefits Estimate	Notes
		Tables G-3 through G-6 in appendix G of the EA; USEPA, 2024e).
The analysis does not take into account population growth and other changes in long-term trends.	Underestimate	The benefits analysis does not reflect the effects of growing population that may benefit from reduction in PFOA/PFOS exposure, which is expected to result in underestimated benefits. The EPA uses present-day information on life expectancy, disease, environmental exposure, and other factors, which are likely to change in the future.
For PWSs with multiple EP, the analysis assumes a uniform population distribution across the EP.	Uncertain	Data on the populations served by each EP are not available, and the EPA therefore uniformly distributes system population across EP. Effects of the regulatory alternative may be greater or smaller than estimated, depending on actual populations served by affected EP. For one large system serving more than one million customers the EPA has sufficient data on EP flow to proportionally assign effected populations.
The EPA does not characterize uncertainty associated with the Value of Statistical Life reference value or Value of Statistical Life elasticity	Uncertain	The EPA did not quantitatively characterize the uncertainty for the Value of Statistical Life reference value and income elasticity. Because the economic value of avoided premature mortality comprises most of the overall benefits estimate, not considering uncertainty surrounding the Value of Statistical Life is a limitation.
Process wastes not classified as hazardous	Underestimate	The national EA reflects the assumption that PFAS-contaminated wastes are not considered RCRA regulatory or characteristic hazardous wastes. The EPA acknowledges that if Federal authorities later determine that PFAS-contaminated wastes require handling as hazardous wastes, there will be additional benefits to public health and the environment from reduced exposures to PFAS that have not been quantified as part of this analysis.

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G. Nonquantifiable Benefits of PFOA and PFOS Exposure Reduction

In this section, the EPA qualitatively discusses the potential health benefits resulting from reduced exposure to PFOA and PFOS in drinking water. These nonquantifiable benefits are expected to be realized as avoided adverse health effects as a result of the final NPDWR, in addition to the benefits that the EPA has quantified, because of their known toxicity and additive health concerns as well as occurrence and likely co-occurrence in drinking water. The EPA anticipates additional benefits associated with developmental, cardiovascular, liver, immune, endocrine, metabolic, reproductive, musculoskeletal, and carcinogenic effects beyond those benefits that the EPA has quantified. The evidence for these adverse health effects is briefly summarized here.

The EPA identified a wide range of potential health effects associated with exposure to PFOA and PFOS using five comprehensive Federal Government health effects assessments that summarize the recent literature on PFAS (mainly PFOA and PFOS, although many of the same health effects have been observed for the other PFAS in this rule) exposure and its health impacts: the EPA's HESDs for PFOA and PFOS, hereafter referred to as the EPA HESDs (USEPA, 2016c; USEPA, 2016d); the EPA's Final Human Health Toxicity Assessments for PFOA and

PFOS (USEPA, 2024c; USEPA, 2024d); and the U.S. Department of Health and Human Services (HHS) ATSDR *Toxicological Profile for Perfluoroalkyls* (ATSDR, 2021). Each source presents comprehensive literature reviews on adverse health effects associated with PFOA and PFOS. The EPA notes that NASEM also published a report which includes a review of the adverse health effects for numerous PFAS (NASEM, 2022). That document is included in the docket for this final rule.

The most recent literature reviews on PFAS exposures and health impacts, which are included in the EPA's *Final Human Health Toxicity Assessments for PFOA and PFOS* (USEPA, 2024c; USEPA, 2024d), describe the weight of evidence supporting PFOA and PFOS associations with health outcomes as either demonstrative, indicative (likely), suggestive, inadequate, or strong evidence supportive of no effect according to the evidence integration judgments outlined in the *ORD Staff Handbook for Developing IRIS Assessments* (USEPA, 2022f; USEPA, 2024c; USEPA, 2024d). For the purposes of the reviews conducted to develop the *Final Human Health Toxicity Assessments for PFOA and PFOS,* an association is deemed demonstrative when there is a strong evidence base demonstrating that the chemical exposure causes a health effect in humans. The association is deemed indicative (likely) when the evidence base indicates that the chemical exposure likely causes a health effect in humans, although there might be outstanding questions or limitations that remain, and the evidence is insufficient for the higher conclusion level. The association is suggestive if the evidence base suggests that the chemical exposure might cause a health effect in humans, but there are very few studies that contributed to the evaluation, the evidence is very weak or conflicting, or the methodological conduct of the studies is poor. The association is inadequate if there is a lack of information or an inability to interpret the available evidence (*e.g.,* findings across studies). The association supports no effect when extensive evidence across a range of populations and exposure levels has identified no effects/associations. Note that the EPA considered information available as of September 2023 for the analyses presented herein.

Developmental effects: Exposure to PFOA and PFOS is linked to developmental effects including but not limited to the infant birth weight effects that the EPA quantified. Other developmental effects include small for gestational age (SGA), birth length, head circumference at birth, and other effects (Verner et al., 2015; Negri et al., 2017; ATSDR, 2021; Waterfield et al., 2020; USEPA, 2016c; USEPA, 2016d; USEPA, 2024c; USEPA, 2024d). SGA is a developmental health outcome of interest when studying potential effects of PFOA/PFOS exposure because SGA infants have increased health risks during pregnancy and delivery as well as post-delivery (Osuchukwu and Reed, 2022). The majority of epidemiology studies indicated increased risk of SGA with PFOA/PFOS exposure, although some studies reported null results (USEPA, 2024c; USEPA, 2024d). For instance, some studies suggested a potentially positive association between PFOA exposure and SGA (Govarts et al., 2018; Lauritzen et al., 2017; Wang et al., 2016; Souza et al., 2020; Wikström et al., 2020; Chang et al., 2022; USEPA, 2024c). In addition to decreases in offspring weight, toxicology studies on PFOA and PFOS exposures in rodents demonstrated relationships with multiple other developmental toxicity endpoints, including increased offspring mortality, decreased maternal body weight and body weight change, skeletal and soft tissue effects, and delayed eye-opening (USEPA, 2024c; USEPA, 2024d). For additional details on developmental studies and their individual outcomes, see chapter 3.4.4 (*Developmental*) in USEPA (2024c) and USEPA (2024d).

Cardiovascular effects: In addition to the CVD effects that the EPA quantified associated with changes in TC and blood pressure from exposure to PFOA and PFOS (see section 6.2 of USEPA (2024g)), available evidence suggests an association between exposure to PFOA and PFOS and increased low-density lipoprotein cholesterol (LDLC) (ATSDR, 2021; USEPA, 2024c; USEPA, 2024d). High levels of LDLC are known as the 'bad' cholesterol because it can lead to the buildup of cholesterol in the arteries, which can raise the risk of heart disease and stroke. Epidemiology studies showed a positive association between PFOA or PFOS exposure and LDLC levels in adults and children (USEPA, 2024c; USEPA, 2024d). In particular, the evidence suggested positive associations between serum PFOA and PFOS levels and LDLC levels in adolescents ages 12–18, while positive associations between serum levels and LDLC levels in younger children were observed only for PFOA (ATSDR, 2021). Additionally, available evidence supports a relatively consistent positive association between PFOA or PFOS and low-density lipoprotein (LDL) in adults, especially those who are obese or prediabetic. Associations with other lipoprotein cholesterol known to increase cardiovascular risks were also positive, which increased confidence in the findings for LDLC. Available evidence regarding the impact of PFOA and PFOS exposure on pregnant women was too limited for the EPA to determine an association (ATSDR, 2021; USEPA, 2024c; USEPA, 2024d). Toxicology studies generally reported alterations in serum lipid levels in mice and rats following oral exposure to PFOA (USEPA, 2024d) or PFOS (USEPA, 2024c), indicating a disruption in lipid metabolism, which is coherent with effects observed in humans. For additional details on LDLC studies and their individual outcomes, see chapter 3.4.3 (*Cardiovascular*) in USEPA (2024c) and USEPA (2024d).

Liver effects: Several biomarkers can be used clinically to diagnose liver diseases, including alanine aminotransferase (ALT). Serum ALT measures are considered a reliable indicator of impaired liver function because increased serum ALT is indicative of leakage of ALT from damaged hepatocytes (Boone et al., 2005; Z. Liu et al., 2014; USEPA, 2002d). Additionally, evidence from both human epidemiological and animal toxicological studies indicates that increased serum ALT is associated with liver disease (Ioannou et al., 2006a; Ioannou et al., 2006b; Kwo et al., 2017; Roth et al., 2021). Human epidemiological studies have demonstrated that even low magnitude increases in serum ALT can be clinically significant (Mathiesen et al., 1999; Park et al., 2019). Additionally, numerous studies have demonstrated an association between elevated ALT and liver-related mortality (reviewed by Kwo et al., 2017). Furthermore, the American Association for the Study of Liver Diseases (AASLD) recognizes serum ALT as an indicator of overall human health and mortality (Kim et al., 2008). Epidemiology data provides consistent evidence of a positive association between PFOS/PFOA exposure and ALT levels in adults (ATSDR, 2021; USEPA, 2024c; USEPA, 2024d). Studies of adults showed consistent evidence of a positive association between PFOA exposure and elevated ALT levels at both high exposure levels and exposure levels typical of the general population (USEPA, 2024c). There is also consistent epidemiology evidence of associations between PFOS and elevated ALT levels. A limited number of studies reported inconsistent evidence on whether PFOA/PFOS exposure is associated with increased risk of liver disease (USEPA, 2024d). It is also important to note that while evaluation of direct liver damage is possible in animal studies, it is difficult to obtain biopsy-confirmed histological data in humans. Therefore, liver injury is typically assessed using serum biomarkers of hepatotoxicity (Costello et al., 2022). Associations between PFOA/PFOS exposure and ALT levels in children were less consistent than in adults (USEPA, 2024c; USEPA, 2024d).

PFOA toxicology studies showed increases in ALT and other serum liver enzymes across multiple species, sexes, and exposure paradigms (USEPA, 2024c). Toxicology studies on the impact of PFOS exposure on ALT also reported increases in ALT and other serum liver enzyme levels in rodents, though these increases were modest (USEPA, 2024d). Several studies in animals also reported increases in the incidence of liver lesions or cellular alterations, such as hepatocellular cell death (USEPA, 2024c; USEPA, 2024d). For additional details on the ALT studies and their individual outcomes, see section 3.4.1 (Hepatic) in USEPA (2024c) and USEPA (2024d).

Immune effects: Proper antibody response helps maintain the immune system by recognizing and responding to antigens. The available evidence indicates a relationship between PFOA exposure and immunosuppression; epidemiology studies showed suppression of at least one measure of the antibody response for tetanus and diphtheria among people with higher prenatal and childhood serum concentrations of PFOA (ATSDR, 2021; USEPA, 2024c). Data reporting on associations between PFOA exposure and antibody response to vaccinations other than tetanus and diphtheria (*i.e.,* rubella and hand, foot, and mouth disease) are limited but supportive of associations between PFOA and decreased immune response in children (USEPA, 2024c). Available studies supported an association between PFOS exposure and immunosuppression in children, where increased PFOS serum levels were associated with decreased antibody production in response to tetanus, diphtheria, and rubella vaccinations (USEPA, 2024d). Studies reporting associations between PFOA or PFOS exposure and immunosuppression in adults are less consistent, though this may be due to a lack of high confidence data (USEPA, 2024c; USEPA, 2024d). Toxicology evidence suggested that PFOA and PFOS exposure results in effects similarly indicating immune suppression, such as reduced response of immune cells to challenges (*e.g.,* reduced natural killer cell activity and immunoglobulin production) (USEPA, 2024c; USEPA, 2024d). For additional details on immune studies and their individual outcomes, see section 3.4.2 (*Immune*) in USEPA (2024c) and USEPA (2024d).

Endocrine effects: Elevated circulating thyroid hormone levels can accelerate metabolism and cause irregular heartbeat; low levels of thyroid hormones can cause neurodevelopmental effects, tiredness, weight gain, and increased susceptibility to the common cold. There is suggestive evidence of a positive association between PFOA/PFOS exposure and thyroid hormone disruption (ATSDR, 2021; USEPA, 2024c; USEPA, 2024d). Epidemiology studies reported inconsistent evidence regarding associations between PFOA and PFOS exposure and general endocrine outcomes, such as thyroid disease, hypothyroidism, and hypothyroxinemia (USEPA, 2024c; USEPA, 2024d). However, for PFOA, epidemiological studies reported suggestive evidence of positive associations for serum levels of thyroid stimulating hormone (TSH) and the thyroid hormone triiodothyronine (T3) in adults, and the thyroid hormone thyroxine (T4) in children (USEPA, 2024c; USEPA, 2024d). For PFOS, epidemiological studies reported suggestive evidence of positive associations for TSH in adults, positive associations for T3 in children, and inverse associations for T4 in children (USEPA, 2024d). Toxicology studies indicated that PFOA and PFOS exposure leads to decreases in serum thyroid hormone levels[26] and adverse effects to the endocrine system (ATSDR, 2021; USEPA, 2024c; USEPA, 2024d; USEPA, 2024h). Overall, changes in serum thyroid hormone levels in animals indicate PFOS and PFOA toxicity potentially relevant to humans (USEPA, 2024c; USEPA, 2024d). For additional details on endocrine effects studies and their individual outcomes, see appendix C.2 (*Endocrine*) in USEPA (2024h) and USEPA (2024i).

Metabolic effects: Leptin is a hormone that, along with adiponectin, can be a marker of adipose tissue dysfunction. Chronic high levels of leptin lead to leptin resistance that mirrors many of the characteristics associated with diet-induced obesity, including reduced leptin receptors and diminished signaling. Therefore, high leptin levels are associated with higher body fat mass, a larger size of individual fat cells, overeating, and inflammation (*e.g.,* of adipose tissue, the hypothalamus, blood vessels, and other areas). Evidence suggests an association between PFOA exposure and leptin levels in the general adult population (ATSDR, 2021; USEPA, 2024c). Based on a review of human epidemiology studies, evidence of associations between PFOS and metabolic outcomes appears inconsistent, but in some studies, positive associations were observed between PFOS exposure and leptin levels (USEPA, 2024d). Studies examining newborn leptin levels did not find associations with maternal PFOA levels (ATSDR, 2021). Maternal PFOS levels were also not associated with alterations in leptin levels (ATSDR, 2021). For additional details on metabolic effect studies and their individual outcomes, see appendix C.3 (*Metabolic/Systemic*) in USEPA (2024h) and USEPA (2024i).

Reproductive effects: Studies of the reproductive effects from PFOA/PFOS exposure have focused on associations between exposure to these contaminants and increased risk of gestational hypertension and preeclampsia in pregnant women (ATSDR, 2021; USEPA, 2024c; USEPA, 2024d). Gestational hypertension (high blood pressure during pregnancy) can lead to fetal problems such as poor growth and stillbirth. Preeclampsia—instances of gestational hypertension where the mother also has increased levels of protein in her urine—can similarly pose significant risks to both the fetus and mother. Risks to the fetus include impaired fetal growth due to the lack of oxygen and nutrients, stillbirth, preterm birth, and infant death (NIH, 2017). Even if born full term, the infant may be at risk for later problems such as diabetes, high blood pressure, and congestive heart failure. Effects of preeclampsia on the mother may include kidney and liver damage, blood clotting problems, brain injury, fluid on the lungs, seizures, and mortality (NIH, 2018). The epidemiology evidence yields mixed (positive and null) associations, with some suggestive evidence supporting positive associations between PFOA/PFOS exposure and both preeclampsia and gestational hypertension (ATSDR, 2021; USEPA, 2024c; USEPA, 2024d). For additional details on reproductive effects studies and their individual

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[26] Decreased thyroid hormone levels are associated with effects such as changes in thyroid and adrenal gland weight, hormone fluctuations, and organ histopathology, as well as adverse neurodevelopmental outcomes (ATSDR, 2021; USEPA, 2024c).

outcomes, see appendix C.1 (*Reproductive*) in USEPA (2024h) and USEPA (2024i).

Musculoskeletal effects: Adverse musculoskeletal effects such as osteoarthritis and decreased bone mineral density impact bone integrity and cause bones to become brittle and more prone to fracture. The available epidemiology evidence suggests that PFOA exposure may be linked to decreased bone mineral density, bone mineral density relative to bone area, height in adolescence, osteoporosis, and osteoarthritis (ATSDR, 2021; USEPA, 2024c). Some studies found that PFOA/PFOS exposure was linked to osteoarthritis, in particular among women under 50 years of age (ATSDR, 2021). There is limited evidence from studies pointing to effects of PFOS on skeletal size (height), lean body mass, and osteoarthritis (USEPA, 2024d). Evidence from some studies suggests that PFOS exposure has a harmful effect on bone health, particularly measures of bone mineral density, with greater statistically significance of effects occurring among females (USEPA, 2024d). However, other reviews reported mixed findings on the effects of PFOS exposure including decreased risk of osteoarthritis, increased risk for some demographic subgroups, or no association (ATSDR, 2021). For additional details on musculoskeletal effects studies and their individual outcomes, see appendix C.8 (*Musculoskeletal*) in USEPA (2024h) and USEPA (2024i).

Cancer Effects: In the EPA's *Final Human Health Toxicity Assessment for PFOA,* the agency evaluates the evidence for carcinogenicity of PFOA that has been documented in both epidemiological and animal toxicity studies (USEPA, 2024c; USEPA, 2024j). The evidence in epidemiological studies is primarily based on the incidence of kidney and testicular cancer, as well as potential incidence of breast cancer in genetically susceptible subpopulations or for particular breast cancer types. Other cancer types have been observed in humans, although the evidence for these is generally limited to low confidence studies. The evidence of carcinogenicity in animal models is provided in three chronic oral animal bioassays in Sprague-Dawley rats which identified neoplastic lesions of the liver, pancreas, and testes (USEPA, 2024c; USEPA, 2024j). The EPA determined that PFOA is *Likely to Be Carcinogenic to Humans,* as ''the evidence is adequate to demonstrate carcinogenic potential to humans but does not reach the weight of evidence for the descriptor Carcinogenic to Humans.'' This determination is based on the evidence of kidney and testicular cancer in humans and LCTs, PACTs, and hepatocellular adenomas in rats (USEPA, 2024c; USEPA, 2024j). The EPA's benefits analysis for avoided RCC cases from reduced PFOA exposure is discussed in section XII.E of this preamble and in section 6.6 of USEPA (2024g).

In the EPA's *Final Human Health Toxicity Assessment for PFOS,* the agency evaluates the evidence for carcinogenicity of PFOS and found that several epidemiological studies and a chronic cancer bioassay comprise the evidence database for the carcinogenicity of PFOS (USEPA, 2024d; USEPA 2024j). The available epidemiology studies report elevated risk of liver cancer, consistent with increased incidence of liver tumors reported in male and female rats. There is also mixed but plausible evidence of bladder, prostate, kidney, and breast cancers in humans. The animal chronic cancer bioassay study also provides evidence of increased incidence of pancreatic islet cell tumors in male rats. The EPA reviewed the weight of the evidence and determined that PFOS is *Likely to Be Carcinogenic to Humans,* as ''the evidence is adequate to demonstrate carcinogenic potential to humans but does not reach the weight of evidence for the descriptor Carcinogenic to Humans.'' The EPA's national-level benefits sensitivity analysis for avoided liver cancer cases from reduced PFOS exposure is detailed in appendix O of the EA.

The EPA anticipates there are additional nonquantifiable benefits related to potential testicular, bladder, prostate, and breast cancer effects summarized above. Benefits associated with avoiding cancer cases not quantified in the EPA's analysis could be substantial. For example, a study by Obsekov et al. (2023) reports the number of breast cancer cases attributable to PFAS exposure ranges from 421 to 3,095 annually, with an estimated direct cost of 6-month treatment ranging from $27.1 to $198.4 million per year ($2022). This study also finds that approximately 5 (0.076 percent) annual testicular cases are attributable to PFOA exposure with an estimated direct cost of treatment of $173,450 per year ($2022). Although the methods used by Obsekov et al. (2023) differ from those used to support the national quantified benefits of the rule, the information provided in the study is helpful in portraying the costs of cancers that are associated with PFAS exposures. For additional details on cancer studies and their individual outcomes, see chapter 3.5 (*Cancer*) in USEPA (2024c) and USEPA (2024d).

After assessing available health and economic information, the EPA was unable to quantify the benefits of avoided health effects discussed above. The agency prioritized health endpoints with the strongest weight of evidence conclusions and readily available data for monetization, namely cardiovascular effects, developmental effects, and carcinogenic effects. Several other health endpoints that had indicative or suggestive evidence of associations with exposure to PFOA and PFOS have not been selected for the EA:

- While immune effects had indicative evidence of associations with exposure to PFOA and PFOS, the EPA did not identify the necessary information to connect the measured biomarker responses (*i.e.,* decrease in antibodies) to a disease that could be valued in the EA;
- Evidence indicates associations between PFOA and PFOS exposure and hepatic effects, such as increases in ALT. While increased ALT is considered an adverse effect, ALT can be one of several contributors to a variety of diseases, including liver disease, and it is difficult to therefore quantify the relationship between this biomarker and a disease that can be monetized. Similar challenges with the biomarkers representing metabolic effects (*i.e.,* leptin) and musculoskeletal effects (*i.e.,* bone density) prevented economic analysis of these endpoints;
- There is evidence of association between exposure to PFOA and testicular cancer in human and animal studies; however, the available slope factor in rats implied small changes in the risk of this endpoint. Because testicular cancer is rarely fatal and the Value of Statistical Life is the driver of economic benefits evaluated in the EA, the benefit of decreased testicular cancer expected with this rule was smaller in comparison and not quantified;
- There is evidence of association between exposure to PFOS and hepatic carcinogenicity in human and animal studies. The EPA quantified benefits associated with reduced liver cancer cases and deaths as part of a sensitivity analysis for the final rule in response to public comments received on the proposed rule requesting that the EPA quantify additional health benefits (see appendix O of the EA (USEPA, 2024e));
- Finally, other health endpoints, such as SGA and LDLC effects, were not modeled in the EA because they overlap with effects that the EPA did model. More specifically, SGA infants are often born with decreased birth weight or

receive similar care to infants born with decreased birth weight. LDLC is a component of TC and could not be modeled separately as the EPA used TC as an input to the ASCVD model to estimate CVD outcomes.

H. Nonquantifiable Benefits of Removal of PFAS Included in the Final Regulation and Co-Removed PFAS

The EPA also qualitatively summarized the potential health benefits resulting from reduced exposure to PFAS other than PFOA and PFOS in drinking water. The final rule and all regulatory alternatives are expected to result in additional benefits that have not been quantified. The final rule will reduce exposure to PFHxS, HFPO–DA, and PFNA to below their individual MCLs. It will also reduce exposure to PFBS to below the Hazard Index MCLG and MCL of 1 when the mixture contains two or more of PFHxS, PFNA, HFPO–DA, and PFBS. Benefits from avoided cases of the adverse health effects discussed in this section are expected from the final rule due to co-occurrence of these contaminants in source waters containing PFOA and/or PFOS, as described in the *Per- and Polyfluoroalkyl Substances (PFAS) Occurrence & Contaminant Background Support Document* (USEPA, 2024b) and part VI of this preamble. In addition, PFAS, including PFHxS, PFNA, HFPO–DA, and PFBS and their mixtures affect common target organs, tissues, or systems to produce dose-additive effects from their co-exposures with each other, as well as PFOA and PFOS (USEPA, 2024a). The EPA expects that compliance actions taken under the final rule will remove additional unregulated co-occurring PFAS contaminants where present because the best available technologies have been demonstrated to co-remove additional PFAS. Treatment responses implemented to reduce PFOA and PFOS exposure under the final rule and Options 1a–c are likely to remove some amount of additional PFAS contaminants where they co-occur.

Ion exchange (IX) and granulated activated carbon (GAC) are effective at removing PFAS; there is generally a linear relationship between PFAS chain length and removal efficiency, shifted by functional group (McCleaf et al., 2017; Sörengård, 2020). Perfluoroalkyl sulfonates (PFSAs), such as PFOS, are removed with greater efficiency than corresponding perfluoroalkyl carboxylates (PFCAs), such as PFOA, of the same carbon backbone length (Appleman et al., 2014; Du et al., 2014; Eschauzier et al., 2012; Ochoa-Herrera and Sierra-Alvarez, 2008; Zaggia et al., 2016). Generally, for a given water type and concentration, PFSAs are removed approximately as effectively as PFCAs, which have two additional fully perfluorinated carbons in the carbon backbone. For example, PFHxS (*i.e.,* sulfonic acid with a six-carbon backbone) is removed approximately as well as PFOA (*i.e.,* carboxylic acid with an eight-carbon backbone) and PFHxA (*i.e.,* carboxylic acid with a six-carbon backbone) is removed approximately as well as PFBS (*i.e.,* sulfonic acid with a four-carbon backbone). Further, PFAS compounds with longer carbon chains display lower percentage decreases in average removal efficiency over time (McCleaf et al., 2017).

In cases where the six PFAS included in the final rule occur at concentrations above their respective regulatory standards, there is also an increased probability of co-occurrence of additional unregulated PFAS. Further, as the same technologies also remove other long-chain and higher carbon/higher molecular weight PFAS, the EPA expects that treatment will provide additional public health protection and benefits due to co-removal of unregulated PFAS that may have adverse health effects. While the EPA has not quantified these additional benefits, the agency expects that these important co-removal benefits will further enhance public health protection.

The EPA identified a wide range of potential health effects associated with exposure to PFAS other than PFOA and PFOS using documents that summarize the recent literature on exposure and associated health impacts: the ATSDR's *Toxicological Profile for Perfluoroalkyls* (ATSDR, 2021); the EPA's toxicity assessment of HFPO–DA (USEPA, 2021b); publicly available IRIS assessments for PFBA and PFHxA (USEPA, 2022g; USEPA, 2023p); the EPA's toxicity assessment of PFBS (USEPA, 2021a); and the recent National Academies of Sciences, Engineering, and Medicine *Guidance on PFAS Exposure, Testing, and Clinical Follow-up* (NASEM, 2022). Note that the determinations of associations between PFAS and associated health effects are based on information available as of September 2023.

Developmental effects: Toxicology and/or epidemiology studies observed evidence of associations between birth weight and/or other developmental effects and exposure to PFBA, PFDA, PFHxS, PFHxA, PFNA, HFPO–DA, PFUnA, and PFBS. Specifically, data from toxicology studies support this association for PFBS, PFBA, PFHxA, and HFPO–DA, while both toxicology and epidemiology studies support this association for PFHxS, PFDA, PFUnA, and PFNA (ATSDR, 2021; USEPA, 2021b; USEPA, 2022g; USEPA, 2023e; Wright et al., 2023). In general, epidemiological studies did not find associations between exposure and adverse pregnancy outcomes (miscarriage, preterm birth, or gestational age) for PFNA, PFUnA and PFHxS (ATSDR, 2021; NASEM, 2022). Epidemiological studies support an association between PFNA, PFHxS or PFDA exposure and developmental effects such as decreases in infant birth weight and birth length, small for gestational age and increased risk of low birth weight (Valvi et al., 2017; Bach et al., 2016; Louis et al., 2018; Wright et al., 2023; Manzano-Salgado et al., 2017; Starling et al., 2017). Few epidemiologic studies also indicate that PFDA exposure is associated with developmental effects (Wikström et al., 2020; Valvi et al., 2017; Luo et al., 2021; Yao et al., 2021). The EPA has determined that evidence indicates that exposure to PFBA or PFHxA likely causes developmental effects, based on moderate evidence from animal studies and indeterminate evidence from human studies (USEPA, 2022g; USEPA, 2023p).

Cardiovascular effects: Epidemiology and/or toxicology studies observed evidence of associations between PFNA, PFDA, and PFHxS exposures and effects on total cholesterol, LDLC, and HDLC. Epidemiological studies report consistent associations between PFHxS and total cholesterol in adults (Cakmak et al., 2022; Dunder et al., 2022; Canova et al., 2020; Lin et al., 2019; Liu et al., 2020; Fisher et al., 2013).

In an analysis based on studies published before 2018, evidence for associations between PFNA exposure and serum lipid levels in epidemiology studies was mixed; associations have been observed between serum PFNA levels and total cholesterol in general populations of adults but not in pregnant women, and evidence in children is inconsistent (ATSDR, 2021). Most epidemiology studies did not observe associations between PFNA and LDLC or HDLC. Epidemiological studies report consistent associations between PFDA and effects on total cholesterol in adults (Cakmak et al., 2022; Dunder et al, 2022; Liu et al., 2020; Dong et al., 2019). Positive associations between PFDA and other serum lipids, adiposity, cardiovascular disease, and atherosclerosis were observed in some epidemiology studies, but findings were inconsistent (Huang et al., 2018; Mattsson et al., 2015; Christensen et al., 2016). A single animal study observed decreases in cholesterol and triglyceride levels in rats at PFDA doses above 1.25 mg/kg/d for 28 days (NTP, 2018b). There was no association between PFBA and serum lipids in a single epidemiology study and no animal studies on PFBA evaluated cardiovascular endpoints (USEPA, 2022g).

Other PFAS for which lipid outcomes were examined in toxicology or epidemiology studies showed limited to no evidence of associations. Studies have examined possible associations between various PFAS and blood pressure in humans or heart histopathology in animals. Epidemiological studies report positive associations between PFHxS and hypertension in adolescents and young adults (Averina et al., 2021; Li et al., 2021; Pitter et al., 2020), but not in other adults (Lin et al., 2020; Chen et al., 2019; Christensen et al., 2018; Liu et al., 2018; Bao et al., 2017; Christensen et al., 2016) or children (Papadopoulou et al., 2021; Khalil et al., 2018; Manzano-Salgado et al., 2017). No evidence was observed of associations between PFHxS and cardiovascular diseases (Huang et al., 2018; Mattsson et al., 2015). Overall, studies did not find likely evidence of cardiovascular effects for other PFAS except for PFOA and PFOS (USEPA, 2024c; USEPA, 2024d).

Hepatic effects: Toxicology and/or epidemiology studies have reported associations between exposure to PFAS (PFBA, PFDA, PFUnA, PFDoDA, PFHxA, PFHxS, HFPO–DA, and PFBS) and hepatotoxicity. The results of the animal toxicology studies provide strong evidence that the liver is a sensitive target of PFHxS, PFNA, PFDA, PFUnA, PFBS, PFBA, PFDoDA, HFPO–DA and PFHxA toxicity. Observed effects in rodents include increases in liver weight, hepatocellular hypertrophy, hyperplasia, and necrosis (ATSDR, 2021; USEPA, 2021b; USEPA, 2022g; USEPA, 2023p). Increases in serum enzymes (such as ALT) and decreases in serum bilirubin were observed in several epidemiological studies of PFNA and PFDA (Nian et al., 2019; Jain and Ducatman, 2019; Liu et al., 2022; Cakmak et al., 2022). Associations between exposure to PFHxS and effects on serum hepatic enzymes are less consistent (Cakmak et al., 2022; Liu et al., 2022; Jain and Ducatman, 2019; Salihovic et al., 2018; Gleason et al., 2015). Mixed effects were observed for serum liver enzymes in epidemiological studies for PFNA (ATSDR, 2021).

Immune effects: Epidemiology studies have reported evidence of associations between PFDA or PFHxS exposure and antibody response to tetanus or diphtheria (Grandjean et al., 2012; Grandjean et al., 2017a; Grandjean et al., 2017b; Budtz-Jørgensen and Grandjean, 2018). There is also some limited evidence for decreased antibody response for PFNA, PFUnA, and PFDoDA, although there were notable inconsistencies across studies examining associations for these compounds (ATSDR, 2021). There is limited evidence for associations between PFHxS, PFNA, PFDA, PFBS, and PFDoDA and increased risk of asthma due to the small number of studies evaluating the outcome and/or inconsistent study results (ATSDR, 2021). The small number of studies investigating immunotoxicity in humans following exposure to PFHpA and PFHxA did not find associations (ATSDR, 2021; USEPA, 2023p; NASEM, 2022). Toxicology studies have reported evidence of associations between HFPO–DA exposure and effects on various immune-related endpoints in animals (ATSDR, 2021; USEPA, 2021b). No laboratory animal studies were identified for PFUnA, PFHpA, PFDoDA, or FOSA. A small number of toxicology studies evaluated the immunotoxicity of other perfluoroalkyls and most did not evaluate immune function. No alterations in spleen or thymus organ weights or morphology were observed in studies on PFHxS and PFBA. A study on PFNA found decreases in spleen and thymus weights and alterations in splenic lymphocyte phenotypes (ATSDR, 2021). Changes in spleen and thymus weights were reported in female mice and male/female rats in two 28-day gavage studies of PFDA, although the direction and dose-dependency of these changes in rats was inconsistent across studies (Frawley et al., 2018; NTP, 2018b).

COVID–19: A cross-sectional study in Denmark (Grandjean et al., 2020) showed that PFBA exposure was associated with increasing severity of COVID–19, with an OR of 1.77 (95% CI: 1.09, 2.87) after adjustment for age, sex, sampling site, and interval between blood sampling and diagnosis. A case-control study showed increased risk of COVID–19 infection with high urinary PFAS (including PFOA, PFOS, PFHxA, PFHpA, PFHxS, PFNA, PFBS, PFDA, PFUnA, PFDoA, PFTrDA, PFTeDA) levels (Ji et al., 2021). Adjusted odds ratios were 1.94 (95% CI: 1.39, 2.96) for PFOS, 2.73 (95% CI: 1.71, 4.55) for PFOA, and 2.82 (95% CI: 1.97–3.51) for total PFAS (sum of 12 PFAS), while other PFAS were not significantly associated with COVID–19 susceptibility after adjusting for confounders. In a spatial ecological analysis, Catelan et al. (2021) showed higher mortality risk for COVID–19 in a population heavily exposed to PFAS (including PFOA, PFOS, PFHxS, PFBS, PFBA, PFPeA, PFHxA, and PFHpA) via drinking water. Overall, results suggested a general immunosuppressive effect of PFAS and/or increased COVID–19 respiratory toxicity due to a concentration of PFBA in the lungs. Although these studies provide a suggestion of possible associations, the body of evidence does not permit conclusions about the relationship between COVID–19 infection, severity, or mortality, and exposures to PFAS.

In addition to the adverse health effects listed above, there was little or no evidence that exposure to the various PFAS is associated with the additional health effects summarized in this section.

Endocrine effects: Epidemiology studies have observed associations between serum PFHxS, PFNA, PFDA, and PFUnA and effects on thyroid stimulating hormone (TSH), triiodothyronine (T3), or thyroxine (T4) levels in serum or thyroid disease; however, there are notable inconsistencies across the studies identified in the available reports (ATSDR, 2021; NASEM, 2022). Toxicology studies have reported consistent associations between exposure to PFHxS, PFBA, PFHxA, and PFBS and effects on thyroid hormones, thyroid organ weight, and thyroid histopathology in animals; the endocrine system was a notable target of PFBS and PFHxS toxicity (ATSDR, 2021; USEPA, 2021a; USEPA, 2022g; USEPA, 2023p; NTP, 2018b; Ramhøj et al., 2018; Ramhøj et al., 2020; Butenhoff et al., 2009).

Metabolic effects: Epidemiology and toxicology studies have examined possible associations between various PFAS and metabolic effects, including leptin, body weight, or body fat in humans or animals (ATSDR, 2021). Exposure to PFDA has been associated with an increase in adiposity in adults (Blake et al., 2018; Christensen et al., 2018; Liu et al., 2018). However, evidence of associations was not suggestive or likely for any PFAS in this summary except for PFOA and PFOS (USEPA, 2024c; USEPA, 2024d). Evidence for changes such as maternal body weight gain, pup body weight, or other developmentally focused weight outcomes is strong but is considered under the Developmental effects category (ATSDR, 2021; NASEM, 2022).

Renal effects: A small number of epidemiology studies with inconsistent results evaluated possible associations between PFHxS, PFNA, PFDA, PFBS, PFDoDA, or PFHxA and renal functions (including estimated glomerular filtration rate and increases in uric acid levels) (ATSDR, 2021; NASEM 2022; USEPA, 2023p). Toxicology studies have not observed impaired renal function or morphological damage following exposure to PFHxS, PFDA, PFUnA, PFBS, PFBA, PFDoDA, or PFHxA (ATSDR, 2021). Associations with kidney weight in animals were observed for PFBS and HFPO–DA and was a notable target for PFBS toxicity (ATSDR, 2021; USEPA, 2021a; USEPA, 2021b; USEPA, 2023p).

Reproductive effects: A small number of epidemiology studies with inconsistent results evaluated possible associations between reproductive hormone levels and PFHxS, PFNA, PFDA, PFUnA, PFDoDA, or PFHxA. Some associations between PFAS (PFHxS, PFHxA, PFNA, PFDA) exposures and sperm parameters have been observed, but often only one sperm parameter was altered. While there is suggestive evidence of an association between PFHxS or PFNA exposure and an increased risk of early menopause, this may be due to reverse causation since an earlier onset of menopause would result in a decrease in the removal of PFAS in menstrual blood. Epidemiological studies provide mixed evidence of impaired fertility (increased risks of longer time to pregnancy and infertility), with some evidence for PFHxS, PFNA, PFHpA, and PFBS but the results are inconsistent across studies or were only based on one study (ATSDR, 2021; Bach et al., 2018; Vélez et al., 2015). Toxicology studies have evaluated the potential histological alterations in reproductive tissues, alterations in reproductive hormones, and impaired reproductive functions. No effect on fertility was observed for PFBS and PFDoDA, and no histological alterations were observed for PFBS and PFBA. One study found alterations in sperm parameters and decreases in fertility in mice exposed to PFNA, and one study for PFDoDA observed ultrastructural alterations in the testes (ATSDR, 2021). Decreased uterine weights, changes in hormone levels, and increased time spent in diestrus were observed in studies of PFDA or PFHxS exposures (NTP, 2018b; Yin et al., 2021).

Musculoskeletal effects: Epidemiology studies observed evidence of associations between PFNA and PFHxS and musculoskeletal effects including osteoarthritis and bone mineral density, but data are limited to two studies (ATSDR, 2021; Khalil et al., 2016; Khalil et al., 2018). Toxicology studies reported no morphological alterations in bone or skeletal muscle in animals exposed to PFBA, PFDA, PFHxA, PFHxS, or PFBS, but evidence is based on a very small number of studies (NTP, 2018b; ATSDR, 2021; USEPA, 2022g; USEPA, 2023p).

Hematological effects: A single uninformative epidemiological study reported on blood counts in pregnant women exposed to PFHxA (USEPA, 2023p). Epidemiological data were not identified for the other PFAS (ATSDR, 2021). A limited number of toxicology studies observed alterations in hematological indices following exposure to relatively high doses of PFHxS, PFDA, PFUnA, PFBS, PFBA, or PFDoDA (ATSDR, 2021; USEPA, 2022g; NTP, 2018b; 3M Company, 2000; Frawley et al., 2018). Toxicology studies observed robust evidence of association between PFHxA or HFPO–DA exposure and hematological effects, including decreases in red blood cell (RBC) number, hemoglobin, and percentage of RBCs in the blood (USEPA, 2021b; USEPA, 2023p). A small number of toxicology studies observed slight evidence of associations between exposure to PFHxS, PFDA, or PFBA and decreases in multiple red blood cell parameters and in prothrombin time; however, effects were not consistent (USEPA, 2022g; Butenhoff et al., 2009).

Other non-cancer effects: A limited number of epidemiology and toxicology studies have examined possible associations between various PFAS and dermal, ocular, and other non-cancer effects. However, the evidence does not support associations for any PFAS in this summary except for PFOA and PFOS (ATSDR, 2021; USEPA, 2021a; USEPA, 2023p).

Cancer effects: A small number of epidemiology studies reported limited associations between multiple PFAS (*i.e.,* PFHxS, PFDA, PFUnA, and FOSA) and cancer effects. No consistent associations were observed for breast cancer risk for PFHxS, PFHxA, PFNA, PFHpA, or PFDoDA; increased breast cancer risks were observed for PFDA and FOSA, but this was based on a single study (Bonefeld-Jørgensen et al., 2014), and one study observed non-significant increased risk for breast cancer risk and PFDA (Tsai et al., 2020). Exposure to PFHxS was associated with increased breast cancer risk in one study and with decreased breast cancer risk in two related studies (Bonefeld-Jørgensen et al., 2014; Ghisari et al., 2017; Tsai et al., 2020). No associations between exposure to PFHxS, PFNA, PFDA, or PFUnA and prostate cancer risk were observed. However, among men with a first-degree relative with prostate cancer, associations were observed for PFHxS, PFDA (Hardell et al., 2014), and PFUnA, but not for PFNA (ATSDR, 2021; USEPA, 2022g; USEPA, 2023p). A decreased risk of thyroid cancer was associated with exposure to PFHxS and PFDA in a single study (Liu et al., 2021). Epidemiological studies examining potential cancer effects were not identified for PFBS or PFBA (ATSDR, 2021; USEPA 2022g). No animal studies examined carcinogenicity of PFHxS or PFBA. Aside from a study that suggested an increased incidence of liver tumors in rats exposed to high doses of HFPO–DA, the limited number of available toxicology studies reported no evidence of associations between exposure to other PFAS (*i.e.,* PFDA and PFHxA) and risk of cancer (ATSDR, 2021; USEPA, 2021b; USEPA, 2023p). At this time, there is inadequate information to assess carcinogenic potential for PFAS other than PFOA, PFOS, and HFPO–DA.

I. Benefits Resulting From Disinfection By-Product Co-Removal

As part of its HRRCA, the EPA is directed by SDWA to evaluate quantifiable and nonquantifiable health risk reduction benefits for which there is a factual basis in the rulemaking record to conclude that such benefits are likely to occur from reductions in co-occurring contaminants that may be attributed solely to compliance with the MCL (SDWA 1412(b)(3)(C)(II)). These co-occurring contaminants are expected to include additional PFAS contaminants not directly regulated by the final PFAS NPDWR, co-occurring chemical contaminants such as SOCs, VOCs, and DBP precursors. In this section, the EPA presents a quantified estimate of the reductions in DBP formation potential that are likely to occur as a result of compliance with the final PFAS NPDWR. The methodology detailed here and in section 6.7.1 of USEPA (2024g) to estimate DBP reductions was externally peer reviewed by three experts in GAC treatment for PFAS removal and DBP formation potential (USEPA, 2023m). The external peer reviewers supported the EPA's approach and edits based on their recommendations for clarity and completeness are reflected in the following analysis and discussion.

DBPs are formed when disinfectants react with naturally occurring materials in water. There is a substantial body of literature on DBP precursor occurrence and THM4 formation mechanisms in drinking water treatment. Under the Stage 2 Disinfectants and Disinfection Byproducts Rule (Stage 2 DBP Rule, USEPA, 2006a), the EPA regulates 11 individual DBPs from three subgroups: THM4, HAA5, and two inorganic compounds (bromate and chlorite). The formation of THM4 in a particular drinking water treatment plant is a function of several factors including disinfectant type, disinfectant dose, bromide concentration, organic material type and concentration, temperature, pH, and system residence times. Epidemiology studies have shown that THM4 exposure, a surrogate for chlorinated drinking water, is associated with an increased risk of bladder cancer, among other diseases (Cantor et al., 1998; Cantor et al., 2010; Costet et al., 2011; Beane Freeman et al., 2017; King and Marrett, 1996; Regli et al., 2015; USEPA, 2019d; Villanueva et al., 2004; Villanueva et al., 2006; Villanueva et al., 2007). These studies considered THM4 as surrogate measures for DBPs formed from the use of chlorination that may co-occur. The relationships between exposure to DBPs, specifically THM4 and other halogenated compounds resulting from water chlorination, and bladder cancer are further discussed in section 6.7 of USEPA (2024g). Reductions in exposure to THM4 is expected to yield public health benefits, including a decrease in bladder cancer incidence (Regli et al., 2015). Among other things, Weisman et al. (2022) found that there is even a stronger weight of evidence linking DBPs and bladder cancer since the promulgation of the 2006 Stage 2 DBP regulations (USEPA, 2006a) and publication of Regli et al. (2015). While not the regulated contaminant for this rulemaking, the expected reduction of DBP precursors and subsequent DBPs that result from this rulemaking are anticipated to reduce cancer risk in the U.S. population.

GAC adsorption has been used to remove SOCs, taste and odor compounds, and natural organic matter (NOM) during drinking water treatment (Chowdhury et al., 2013). Recently, many water utilities have installed or are considering installing GAC and/or other advanced technologies as a protective or mitigation measure to remove various contaminants of emerging concern, such as PFAS (Dickenson and Higgins, 2016). Because NOM often exists in a much higher concentration (in mg/L) than trace organics (in µg/L or ng/L) in water, NOM, often measured as TOC, can interfere with the adsorption of trace organics by outcompeting the contaminants for adsorption sites and by general fouling (blockage of adsorption pores) of the GAC.

NOM and inorganic matter are precursors for the formation of THMs and other DBPs when water is disinfected using chlorine and other disinfectants to control microbial contaminants in finished drinking water. Removal of DBP precursors through adsorption onto GAC has been included as a treatment technology for compliance with the existing DBP Rules and is a BAT for the Stage 2 DBP Rule. Dissolved organic matter (DOM) can be removed by GAC through adsorption and biodegradation (Crittenden et al., 1993; Kim et al., 1997; Yapsakli et al., 2010). GAC is well-established for removal of THM and HAA precursors (Cheng et al., 2005; Dastgheib et al., 2004; Iriarte-Velasco et al., 2008; Summers et al., 2013; Cuthbertson et al., 2019; Wang et al., 2019). In addition to removal of organic DBPs, GAC also exhibits some capacity for removal of inorganic DBPs such as bromate and chlorite (Kirisits et al., 2000; Sorlini et al., 2005) and removal of preformed organic DBPs via adsorption and biodegradation (Jiang et al., 2017; Terry and Summers, 2018). Further, GAC may offer limited removal of dissolved organic nitrogen (Chili et al., 2012).

Based on an extensive review of published literature in sampling studies where both contaminant groups (PFAS and DBPs) were sampled, there is limited information about PFAS removal and co-occurring reductions in DBPs, specifically THMs. To help inform its EA, the EPA relied on the DBP Information Collection Rule Treatment Study Database and DBP formation studies to estimate reductions in THM4 (ΔTHM4) that may occur when GAC is used to remove PFAS. Subsequently, these results were compared to THM4 data from PWSs that have detected PFAS and have indicated use of GAC.

The objective of the EPA's co-removal benefits analysis is to determine the reduction in bladder cancer cases associated with the decrease of regulated THM4 in treatment plants due to the installation of GAC for PFAS removal. Evaluation of the expected reductions in bladder cancer risk resulting from treatment of PFAS in drinking water involves five steps:

1. Estimating the number of systems expected to install GAC treatment in compliance with the final PFAS NPDWR and affected population size;
2. Estimating changes in THM4 levels that may occur when GAC is installed for PFAS removal based on influent TOC levels;
3. Estimating changes in the cumulative risk of bladder cancer using an exposure-response function linking lifetime risk of bladder cancer to THM4 concentrations in residential water supply (Regli et al., 2015);
4. Estimating annual changes in the number of bladder cancer cases and excess mortality in the bladder cancer population corresponding to changes in THM4 levels under the regulatory alternative in all populations alive during or born after the start of the evaluation period; and
5. Estimating the economic value of reducing bladder cancer morbidity and mortality from baseline to regulatory alternative levels, using COI measures and the Value of a Statistical Life, respectively.

The EPA expects PWSs that exceed the PFAS MCLs to consider both treatment and nontreatment options to achieve compliance with the drinking water standard. The EPA assumes that the populations served by systems with EP expected to install GAC based on the compliance forecast detailed in section 5.3 of USEPA (2024g) will receive the DBP exposure reduction benefits. The EPA notes that other compliance actions included in the compliance forecast could result in DBP exposure reductions, including installation of RO. However, these compliance actions are not included in the DBP benefits analysis because this DBP exposure reduction function is specific to GAC. Switching water sources may or may not result in DBP exposure reductions, therefore the EPA assumed no additional DBP benefits for an estimated percentage of systems that elect this compliance option. Lastly, the EPA assumed no change in DBP exposure at water systems that install IX, as that treatment technology is not expected to remove a substantial amount of DBP precursors. The EPA also assumed that the PWSs included in this analysis use chlorine only for disinfection and have conventional treatment in place prior to GAC installation.

The EPA used the relationship between median raw water TOC levels and changes in THM4 levels estimated in the 1998 DBP Information Collection Rule to estimate changes in THM4 concentrations in the finished water of PWSs fitted with GAC treatment. For more detail on the approach the EPA used to apply changes in THM4 levels to PWSs treating for PFAS under the final rule, please see section 6.7 of USEPA (2024g).

The EPA models a scenario where reduced exposures to THM4 begin in 2029. Therefore, the EPA assumed that the population affected by reduced THM4 levels resulting from implementation of GAC treatment is exposed to baseline THM4 levels prior to actions to comply with the rule (*i.e.,* prior to 2029) and to reduced THM4 levels from 2029 through 2105. Rather than modeling individual locations (*e.g.,* PWS), the EPA evaluates changes in bladder cancer cases among the aggregate population per treatment scenario and source water type that is expected to install GAC treatment to reduce PFAS levels. Because of this aggregate modeling approach, the EPA used national-level population estimates to distribute the SDWIS populations based on single-year age and sex and to extrapolate the age- and sex-specific populations to future years. Appendix B of USEPA (2024g) provides additional details on estimation of the affected population.

Regli et al. (2015) analyzed the potential lifetime bladder cancer risks associated with increased bromide levels in surface source water resulting in increased THM4 levels in finished water. To account for variable levels of uncertainty across the range of THM4 exposures from the pooled analysis of Villanueva et al. (2004), they derived a weighted mean slope factor from the odds ratios reported in Villanueva et al. (2004). They showed that, while the original analysis deviated from linearity, particularly at low concentrations, the overall pooled exposure-response relationship for THM4 could be well-approximated by a linear slope factor that predicted an incremental lifetime cancer risk of 1 in 10,000 exposed individuals (10^{-4}) per 1 µg/L increase in THM4. The linear slope factor developed by Regli et al. (2015) enables estimation of the changes in the lifetime bladder cancer risk associated with lifetime exposures to reduced THM4 levels. Weisman et al. (2022) applied the dose-response information from Regli et al. (2015) and developed a robust, national-level risk assessment of DBP impacts, where the authors estimated that approximately 8,000 of 79,000 annual U.S. bladder cancer cases are attributable to chlorination DBPs, specifically associated with THM4 concentrations.

The EPA estimated changes in annual bladder cancer cases and annual excess mortality in the bladder cancer population due to estimated reductions in lifetime THM4 exposure using a life table-based approach. This approach was used because (1) annual risk of new bladder cancer should be quantified only among those not already experiencing this chronic condition, and (2) bladder cancer has elevated mortality implications.

The EPA used recurrent life table calculations to estimate a water source type-specific time series of bladder cancer incidence for a population cohort characterized by sex, birth year, and age at the beginning of the PFOA/PFOS evaluation period under the baseline scenario and the GAC regulatory alternative. The estimated risk reduction from lower exposure to DBPs in drinking water was calculated based on changes in THM4 levels used as inputs to the Regli et al. (2015)-based health impact function, described in more detail in section 6.7 of USEPA (2024g). The life table analysis accounts for the gradual changes in lifetime exposures to THM4 following implementation of GAC treatment under the regulatory alternative compared to the baseline. The outputs of the life table calculations are the water source type-specific estimates of the annual change in the number of bladder cancer cases and the annual change in excess bladder cancer population mortality.

The EPA used the Value of a Statistical Life to estimate the benefits of reducing mortality associated with bladder cancer in the affected population. The EPA used the cost of illness-based valuation to estimate the benefits of reducing morbidity associated with bladder cancer. Specifically, the EPA used bladder cancer treatment-related medical care and opportunity cost estimates from Greco et al. (2019). Table 63 shows the original cost of illness estimates from Greco et al. (2019), along with the values updated to $2022 used in this analysis.

The EPA received public comments on the EA for the proposed rule related to the EPA's use of cost of illness information for morbidity valuation. Specifically, a couple of commenters recommended that the EPA use willingness to pay information (instead of cost of illness information) when valuing the costs associated with non-fatal illnesses, stating that willingness to pay information better accounts for lost opportunity costs (*e.g.,* lost productivity and pain and suffering) associated with non-fatal illnesses (USEPA, 2024k). To better account for these opportunity costs, the EPA used recently available willingness to pay values in a sensitivity analysis for morbidity associated with bladder cancer. The sensitivity analysis results show that when willingness to pay values are used in bladder cancer benefits analysis, morbidity benefits are increased by approximately 19.9 percent. See appendix O of the EA for full details and results on the willingness to pay sensitivity analyses.

Table 63: Bladder Cancer Morbidity Valuation

Bladder Cancer Subtype[a]	Type of Cost	Cost in First Year ($2010)[b]	Cost in Subsequent Years ($2010)[b]	Cost in First Year ($2022)[c]	Cost in Subsequent Years ($2022)[c]
Non-invasive	Medical care	9,133	916	$12,851	$1,289
	Opportunity cost	4,572	24	$6,212	$33
	Total cost	13,705	941	$19,062	$1,321
Invasive	Medical care	26,951	2,455	$37,922	$3,454
	Opportunity cost	10,513	77	$14,283	$105
	Total cost	37,463	2,532	$52,205	$3,559

Notes:

[a] The estimates for non-invasive bladder cancer subtype were used to value local, regional, and unstaged bladder cancer morbidity reductions, while the estimates for the invasive bladder cancer subtype were used to value distant bladder cancer morbidity reductions.

[b] The estimates come from Greco et al. (2019).

[c] To adjust for inflation, the EPA used U.S. Bureau of Labor Statistics Consumer Price Index for All Urban Consumers: Medical Care Services in U.S. (City Average).

Tables 64 to 67 presents the estimated changes in non-fatal bladder cancer cases and bladder cancer-related deaths from exposure to THM4 due to implementation of GAC treatment by option. The EPA estimated that, over the evaluation period, the final rule will result in annualized benefits from avoided bladder cancer cases and deaths of $380 million.

Table 64: National Bladder Cancer Benefits, Final Rule (PFOA and PFOS MCLs of 4.0 ng/L each, PFHxS, PFNA, and HFPO-DA MCLs of 10 ng/L, each and Hazard Index of 1) (Million $2022)

Benefits Category	2% Discount Rate		
	5th Percentile[1]	Expected Benefits	95th Percentile[1]
Number of Non-Fatal Bladder Cancer Cases Avoided	5,781.0	7,313.0	8,912.7
Number of Bladder Cancer-Related Deaths Avoided	2,029.6	2,567.8	3,129.9
Total Annualized Bladder Cancer Benefits (Million $2022) [2, 3]	$300.64	$380.41	$463.74

Notes: Quantifiable benefits are increased under final rule table results relative to the other options presented because of modeled PFHxS occurrence, which results in additional quantified benefits from co-removed PFOA and PFOS.

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.J of this preamble and Table 75. This range does not include the uncertainty described in Table 62.

[2] See Table 72 for a list of the nonquantifiable benefits, and the potential direction of impact these benefits would have on the estimated monetized annualized benefits in this table.

[3] When using willingness-to-pay metrics to monetize morbidity benefits, total annualized bladder cancer benefits are increased by $75.87 million.

Table 65: National Bladder Cancer Benefits, Option 1a (PFOA and PFOS MCLs of 4.0 ng/L) (Million $2022)

Benefits Category	2% Discount Rate		
	5th Percentile[1]	**Expected Benefits**	**95th Percentile[1]**
Number of Non-Fatal Bladder Cancer Cases Avoided	5,789.3	7,312.9	8,896.0
Number of Bladder Cancer-Related Deaths Avoided	2,032.5	2,567.8	3,123.2
Total Annualized Bladder Cancer Benefits (Million $2022)[2]	$301.06	$380.41	$462.73

Notes:

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.J of this preamble and Table 75. This range does not include the uncertainty described in Table 62.

[2] See Table 72 for a list of the nonquantifiable benefits, and the potential direction of impact these benefits would have on the estimated monetized annualized benefits in this table.

Table 66: National Bladder Cancer Benefits, Option 1b (PFOA and PFOS MCLs of 5.0 ng/L) (Million $2022)

Benefits Category	2% Discount Rate		
	5th Percentile[1]	**Expected Benefits**	**95th Percentile[1]**
Number of Non-Fatal Bladder Cancer Cases Avoided	4,739.4	6,034.0	7,367.1
Number of Bladder Cancer-Related Deaths Avoided	1,664.0	2,118.7	2,587.1
Total Annualized Bladder Cancer Benefits (Million $2022) [2]	$246.48	$313.88	$383.32

Notes:

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.J of this preamble and Table 75. This range does not include the uncertainty described in Table 62.

[2] See Table 72 for a list of the nonquantifiable benefits, and the potential direction of impact these benefits would have on the estimated monetized annualized benefits in this table.

Table 67: National Bladder Cancer Benefits, Option 1c (PFOA and PFOS MCLs of 10.0 ng/L) (Million $2022)

Benefits Category	2% Discount Rate		
	5th Percentile[1]	Expected Benefits	95th Percentile[1]
Number of Non-Fatal Bladder Cancer Cases Avoided	2,326.9	3,087.9	3,885.3
Number of Bladder Cancer-Related Deaths Avoided	816.8	1,084.3	1,364.3
Total Annualized Bladder Cancer Benefits (Million $2022) [2]	$120.97	$160.62	$202.14

Notes:

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.J of this preamble and Table 75. This range does not include the uncertainty described in Table 62.

[2] See Table 72 for a list of the nonquantifiable benefits, and the potential direction of impact these benefits would have on the estimated monetized annualized benefits in this table.

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J. Comparison of Costs and Benefits

This section provides a comparison of the incremental costs and benefits of the final rule, as described in chapter 7 of the EA. Included here are estimates of total quantified annualized costs and benefits for the final rule and regulatory alternative MCLs under options 1a-1c, as well as considerations for the nonquantifiable costs and benefits. The EPA's determinations as to whether the costs are justified by the benefits must be based on an analysis of both the quantified costs and benefits as well as the nonquantifiable benefits and nonquantifiable costs, per SDWA 1412(b)(3)(C)(I)–(III).

The incremental cost is the difference between quantified costs that will be incurred if the final rule is enacted over current baseline conditions. Incremental benefits reflect the avoided future adverse health outcomes attributable to PFAS reductions and co-removal of additional contaminants due to actions undertaken to comply with the final rule.

Table 68 provides the incremental quantified costs and benefits of the final rule at a 2 percent discount rate in 2022 dollars. The top row shows total monetized annualized costs including total PWS costs and primacy agency costs. The second row shows total monetized annualized benefits including all endpoints that could be quantified and valued. For both, the estimates are the expected (mean) values and the 5th percentile and 95th percentile quantified estimates from the uncertainty distribution. These percentile estimates come from the distributions of annualized costs and annualized benefits generated by the 4,000 iterations of SafeWater MCBC. Therefore, these distributions reflect the joint effect of the multiple sources of variability and uncertainty for quantified costs, quantified benefits, and the baseline uncertainties such as PFAS occurrence, as detailed in sections 5.1.2, 6.1.2, and chapter 4 of the EA, respectively (USEPA, 2024g). For further discussion of the quantified uncertainties in the EA, see section XII.K of this preamble.

The third row shows net quantified benefits (benefits minus costs). The net annual quantified incremental benefits are $760,000. Because of the variation associated with the use of statistical models such as SafeWater MCBC, the modeled quantified net benefits are nearly at parity. The uncertainty range for net benefits is a negative $622 million to $725 million. Additional uncertainties are presented in Table 72.

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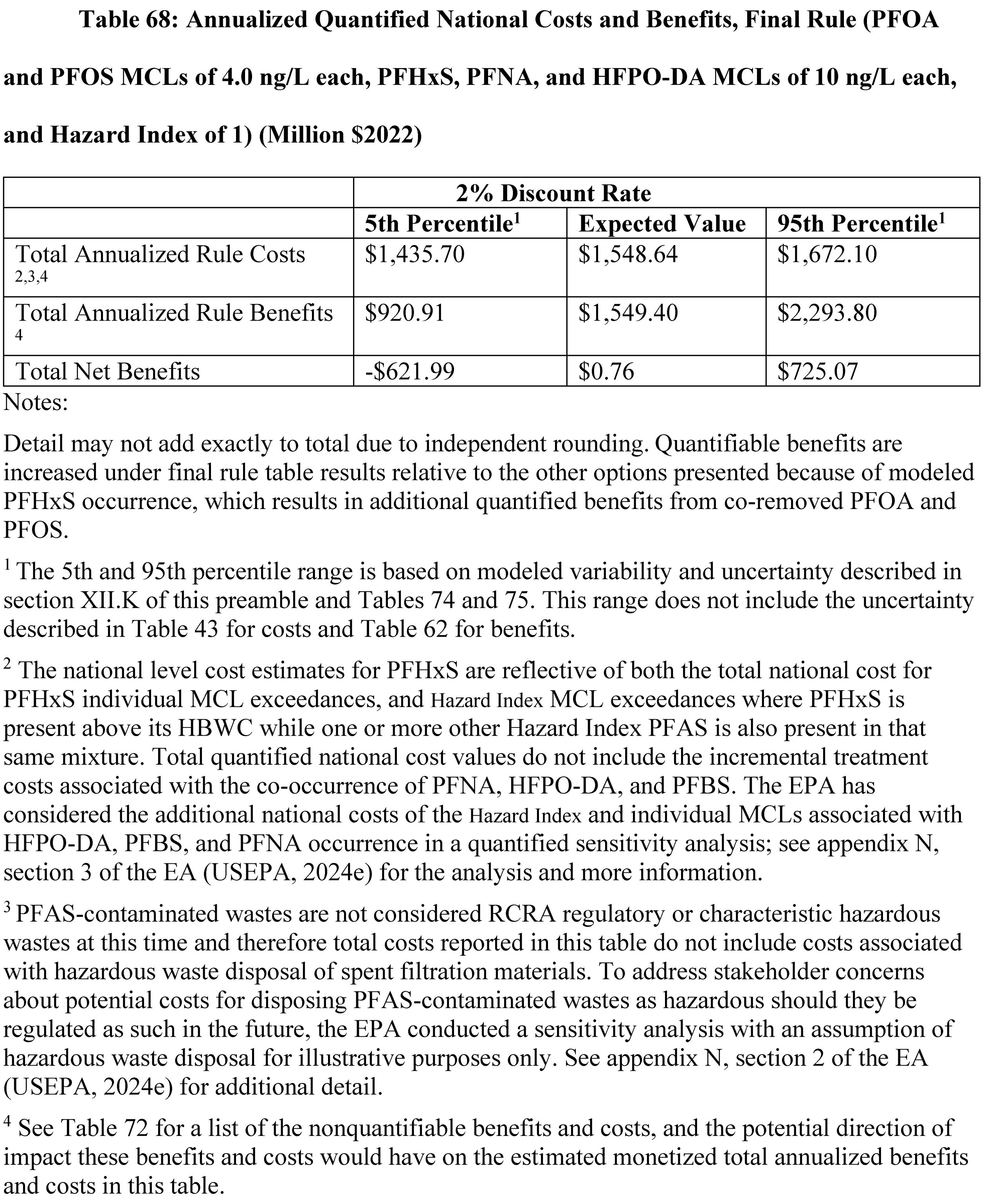

Table 68: Annualized Quantified National Costs and Benefits, Final Rule (PFOA and PFOS MCLs of 4.0 ng/L each, PFHxS, PFNA, and HFPO-DA MCLs of 10 ng/L each, and Hazard Index of 1) (Million $2022)

	2% Discount Rate		
	5th Percentile[1]	**Expected Value**	**95th Percentile[1]**
Total Annualized Rule Costs [2,3,4]	$1,435.70	$1,548.64	$1,672.10
Total Annualized Rule Benefits [4]	$920.91	$1,549.40	$2,293.80
Total Net Benefits	-$621.99	$0.76	$725.07

Notes:

Detail may not add exactly to total due to independent rounding. Quantifiable benefits are increased under final rule table results relative to the other options presented because of modeled PFHxS occurrence, which results in additional quantified benefits from co-removed PFOA and PFOS.

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.K of this preamble and Tables 74 and 75. This range does not include the uncertainty described in Table 43 for costs and Table 62 for benefits.

[2] The national level cost estimates for PFHxS are reflective of both the total national cost for PFHxS individual MCL exceedances, and Hazard Index MCL exceedances where PFHxS is present above its HBWC while one or more other Hazard Index PFAS is also present in that same mixture. Total quantified national cost values do not include the incremental treatment costs associated with the co-occurrence of PFNA, HFPO-DA, and PFBS. The EPA has considered the additional national costs of the Hazard Index and individual MCLs associated with HFPO-DA, PFBS, and PFNA occurrence in a quantified sensitivity analysis; see appendix N, section 3 of the EA (USEPA, 2024e) for the analysis and more information.

[3] PFAS-contaminated wastes are not considered RCRA regulatory or characteristic hazardous wastes at this time and therefore total costs reported in this table do not include costs associated with hazardous waste disposal of spent filtration materials. To address stakeholder concerns about potential costs for disposing PFAS-contaminated wastes as hazardous should they be regulated as such in the future, the EPA conducted a sensitivity analysis with an assumption of hazardous waste disposal for illustrative purposes only. See appendix N, section 2 of the EA (USEPA, 2024e) for additional detail.

[4] See Table 72 for a list of the nonquantifiable benefits and costs, and the potential direction of impact these benefits and costs would have on the estimated monetized total annualized benefits and costs in this table.

Tables 69 to 71 summarize the total annual costs and benefits for options 1a, 1b, and 1c, respectively.

Table 69: Annualized Quantified National Costs and Benefits, Option 1a (PFOA and PFOS MCLs of 4.0 ng/L) (Million $2022)

	2% Discount Rate		
	5th Percentile[1]	**Expected Value**	**95th Percentile[1]**
Total Annualized Rule Costs [2,3]	$1,423.60	$1,537.07	$1,660.30
Total Annualized Rule Benefits [3]	$913.05	$1,542.74	$2,280.10
Total Net Benefits	-$613.79	$5.67	$722.09

Notes:

Detail may not add exactly to total due to independent rounding.

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.K of this preamble and Tables 74 and 75. This range does not include the uncertainty described in Table 43 for costs and Table 62 for benefits.

[2] PFAS-contaminated wastes are not considered RCRA regulatory or characteristic hazardous wastes at this time and therefore total costs reported in this table do not include costs associated with hazardous waste disposal of spent filtration materials. To address stakeholder concerns about potential costs for disposing PFAS-contaminated wastes as hazardous should they be regulated as such in the future, the EPA conducted a sensitivity analysis with an assumption of hazardous waste disposal for illustrative purposes only. See appendix N, section 2 of the EA (USEPA, 2024e) for additional detail.

[3] See Table 72 for a list of the nonquantifiable benefits and costs, and the potential direction of impact these benefits and costs would have on the estimated monetized total annualized benefits and costs in this table.

Table 70: Annualized Quantified National Costs and Benefits, Option 1b (PFOA and PFOS MCLs of 5.0 ng/L) (Million $2022)

	2% Discount Rate		
	5th Percentile[1]	**Expected Value**	**95th Percentile[1]**
Total Annualized Rule Costs [2,3]	$1,102.60	$1,192.13	$1,291.40
Total Annualized Rule Benefits [3]	$768.55	$1,296.84	$1,919.30
Total Net Benefits	-$414.34	$104.71	$710.38

Notes:

Detail may not add exactly to total due to independent rounding.

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.K of this preamble and Tables 74 and 75. This range does not include the uncertainty described in Table 43 for costs and Table 62 for benefits.

[2] PFAS-contaminated wastes are not considered RCRA regulatory or characteristic hazardous wastes at this time and therefore total costs reported in this table do not include costs associated with hazardous waste disposal of spent filtration materials. To address stakeholder concerns about potential costs for disposing PFAS-contaminated wastes as hazardous should they be regulated as such in the future, the EPA conducted a sensitivity analysis with an assumption of hazardous waste disposal for illustrative purposes only. See appendix N, section 2 of the EA (USEPA, 2024e) for additional detail.

[3] See Table 72 for a list of the nonquantifiable benefits and costs, and the potential direction of impact these benefits and costs would have on the estimated monetized total annualized benefits and costs in this table.

Table 71: Annualized Quantified National Costs and Benefits, Option 1c (PFOA and PFOS MCLs of 10.0 ng/L) (Million $2022)

	2% Discount Rate		
	5th Percentile[1]	**Expected Value**	**95th Percentile[1]**
Total Annualized Rule Costs [2,3]	$462.87	$499.29	$540.68
Total Annualized Rule Benefits [3]	$397.28	$664.45	$970.70
Total Net Benefits	-$96.42	$165.16	$468.54

Notes:

Detail may not add exactly to total due to independent rounding.

[1] The 5th and 95th percentile range is based on modeled variability and uncertainty described in section XII.K of this preamble and Tables 74 and 75. This range does not include the uncertainty described in Table 43 for costs and Table 62 for benefits.

[2] PFAS-contaminated wastes are not considered RCRA regulatory or characteristic hazardous wastes at this time and therefore total costs reported in this table do not include costs associated with hazardous waste disposal of spent filtration materials. To address stakeholder concerns about potential costs for disposing PFAS-contaminated wastes as hazardous should they be regulated as such in the future, the EPA conducted a sensitivity analysis with an assumption of hazardous waste disposal for illustrative purposes only. See appendix N, section 2 of the EA (USEPA, 2024e) for additional detail.

[3] See Table 72 for a list of the nonquantifiable benefits and costs, and the potential direction of impact these benefits and costs would have on the estimated monetized total annualized benefits and costs in this table

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The benefit-cost analysis reported dollar figures presented above reflect benefits and costs that could be quantified for each regulatory alternative MCL given the best available scientific data. The EPA notes that these quantified benefits are estimated using a cost-of-illness approach. In the sensitivity analysis, the EPA also calculated quantified benefits using a willingness-to-pay approach instead of cost of illness information, for non-fatal RCC and bladder cancer illnesses. In this case, the estimated expected quantified annualized costs are approximately $1,549 million and the estimated expected quantified annualized benefits increase to approximately $1,632 million, resulting in approximately $84 million in expected annualized net benefits. See appendix O of the EA for further discussion.

The quantified benefit-cost results above are not representative of all benefits and costs anticipated under the final NPDWR. Due to occurrence, health, and economic data limitations, there are several adverse health effects associated with PFAS exposure and costs associated with treatment that the EPA could not estimate quantitatively.

PFAS exposure is associated with a wide range of adverse health effects, including reproductive effects such as decreased fertility; increased high blood pressure in pregnant women; developmental effects or delays in children, including low birth weight, accelerated puberty, bone variations, or behavioral changes; increased risk of some cancers, including prostate, kidney, and testicular cancers; reduced ability of the body's immune system to fight infections, including reduced vaccine response; interference with the body's natural hormones; and increased cholesterol levels and/or risk of obesity. Based on the available data at rule proposal and submitted by public commenters, the EPA is only able to quantify three PFOA- and PFOS-related health endpoints (*i.e.,* changes in birth weight, CVD, and RCC) in the national analysis.

The EPA also evaluated the impacts of PFNA on birth weight and PFOS on liver cancer in quantitative sensitivity analyses (See appendices K and O of USEPA, 2024e, respectively). Those analyses demonstrate that there are potentially significant other quantified benefits not included in the national quantified benefits above: for example, the EPA's quantitative sensitivity analysis for PFNA (found in appendix K of USEPA, 2024e) found that the inclusion of a 1 ng/L PFNA reduction could increase annualized birth weight benefits by a factor of 5.6–7.8 in a model system serving 100,000 people, relative to a scenario that quantified a 1 ng/L reduction in PFOA and a 1 ng/L reduction in PFOS only. In the case of PFOS impacts on liver cancer, the EPA has estimated an expected value of $4.79 million in benefits via the reduction in liver cancer cases anticipated to be realized by the final rule. All regulatory alternatives are expected to produce substantial additional benefits from all the other adverse health effects avoided, but that cannot be quantified at this time. Treatment responses implemented to remove PFOA and PFOS under regulatory alternative MCLs under options 1a-1c are likely to remove some amount of additional PFAS contaminants where they co-occur. Co-occurrence among PFAS compounds has been observed frequently as discussed in the *PFAS Occurrence & Contaminant Background Support Document* (USEPA, 2024b). The final rule is expected to produce the greatest reduction in exposure to PFAS compounds as compared to the three regulatory alternative MCLs because it includes PFHxS, PFNA, HFPO–DA, and PFBS in the regulation. Inclusion of the Hazard Index will trigger more systems to treat (as shown in section 4.4.4 of the EA) and provides enhanced public health protection by ensuring reductions of these additional compounds when present above the Hazard Index of 1. Specifically, as Hazard Index PFAS are reduced, the EPA anticipates additional public health benefits from avoided cardiovascular, developmental, and immune effects. For further discussion of the quantitative and qualitative benefits associated with the final rule, see section 6.2 of the EA.

The EPA also expects that the final rule will result in additional nonquantifiable costs. As noted above, the Hazard Index and individual MCLs are expected to trigger more systems into more frequent monitoring and treatment. In the national cost analysis, the EPA quantified the national treatment and monitoring costs associated with the PFHxS individual MCL and the Hazard Index associated costs based on PFHxS occurrence only. Due to occurrence data limitations, cost estimates for PFNA, PFBS, and HFPO–DA are less precise relative to those for PFOA, PFOS, and PFHxS compounds, and as such, the EPA performed a quantitative sensitivity analysis of the national cost impacts associated with Hazard Index exceedances resulting from PFNA, PFBS, and HFPO–DA and the PNFA and HFPO–DA individual MCLs to understand and consider the potential magnitude of costs associated with treating these three PFAS. The EPA found that in addition to the costs associated with PFHxS exceedances, which are included in the national cost estimate, the Hazard Index and individual MCLs for PFNA and HFPO–DA could cost an additional $82.4 million per year. In cases where these compounds co-occur at locations where PFAS treatment is implemented because of nationally modeled PFOA, PFOS, and PFHxS occurrence, treatment costs are likely to be marginally higher as treatment media estimated bed-life is shortened. In instances where concentrations of PFNA, HFPO–DA, and PFBS are high enough to cause or contribute to a Hazard Index exceedance when the concentrations of PFOA, PFOS, and PFHxS would not have already otherwise triggered treatment, the national modeled costs may be underestimated. If these PFAS occur in isolation at levels that affect treatment decisions, or if these PFAS occur in combination with PFHxS when PFHxS concentrations were otherwise below its respective HBWC in isolation (*i.e.,* less than 10 ng/L) then the quantified costs underestimate the impacts of the final rule. See appendix N.3 of the EA for a sensitivity analysis of additional treatment costs at systems with Hazard Index exceedances (USEPA, 2024e). See appendix N.4 for a sensitivity analysis of the marginal costs of HFPO–DA and PFNA MCLs. For further discussion of how the EPA considered the costs of the five individual MCLs and the HI MCL, see section XII.A.4 of this preamble.

Commenters suggested that another potential source of non-quantified cost comes from the fact that the EPA has proposed designating PFOA and PFOS as CERCLA hazardous substances (USEPA, 2022l). Stakeholders have expressed concern to the EPA that a hazardous substance designation for certain PFAS may limit their disposal options for drinking water treatment residuals (*e.g.,* spent media, concentrated waste streams) and/or potentially increase costs. The designation of PFOA and PFOS as CERCLA hazardous substances would not require waste (*e.g.,* biosolids, treatment residuals, etc.) to be treated in any particular fashion, nor disposed of at any specific particular type of landfill. The designation also would not restrict, change, or recommend any specific activity or type of waste at landfills. In its estimated national costs, the EPA has maintained the assumption that disposal does not have to occur in accordance with hazardous waste standards thus national costs may be underestimated. The EPA has conducted a sensitivity analysis that assumes hazardous waste disposal at all systems treating for PFAS to assess the potential increase in costs (see appendix N of USEPA, 2024e).

Table 72 provides a summary of the likely impact of nonquantifiable benefit-cost categories. In each case, the EPA notes the potential direction of the impact on costs and/or benefits. For example, benefits are underestimated if the PFOA and PFOS reductions result in avoided adverse health outcomes that cannot be quantified and valued. Sections 5.7 and 6.8 of the EA identify the key methodological limitations and the potential effect on the cost or benefit estimates, respectively. Additionally, Table 73 summarizes benefits and costs that are quantified and nonquantifiable under the final rule.

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Table 72: Potential Impact of Nonquantifiable Benefits (B) and Costs (C)

Source	(Final Rule)	Option 1a	Option 1b	Option 1c
Nonquantifiable PFOA and PFOS health endpoints	B: underestimate	B: underestimate	B: underestimate	B: underestimate
Limitations with available occurrence data for PFNA, HFPO-DA, and PFBS	B+C: underestimate	n/a	n/a	n/a
Nonquantifiable HI (PFHxS, PFNA, HFPO-DA, and PFBS) health endpoints	B: underestimate	n/a	n/a	n/a
Limitations with available occurrence data for additional PFAS compounds	B+C: underestimate	B+C: underestimate	B+C: underestimate	B+C: underestimate
Removal of co-occurring non-PFAS contaminants	B+C: underestimate	B+C: underestimate	B+C: underestimate	B+C: underestimate
POU not in compliance forecast	C: overestimate	C: overestimate	C: overestimate	C: overestimate
Unknown future hazardous waste management requirements for PFAS (including HI)	B+C: underestimate	B+C: underestimate	B+C: underestimate	B+C: underestimate

Table 73: Summary of Quantified and Nonquantifiable Benefits and Costs in the National Analysis

Category	Quantified	Non-quantified	Methods (EA Report Section where Analysis is Detailed)
	Costs		
PWS treatment costs [1]	X		Section 5.3.1
PWS sampling costs	X		Section 5.3.2.2
PWS implementation and administration costs	X		Section 5.3.2.1
Primacy agency rule implementation and administration costs	X		Section 5.3.2
Hazardous waste disposal for treatment media		X	Section 5.6
POU not in compliance forecast		X	Section 5.6
	Benefits		
PFOA and PFOS birth weight effects	X		Section 6.4
PFOA and PFOS cardiovascular effects	X		Section 6.5
PFOA and PFOS RCC	X		Section 6.6
Health effects associated with DBPs, specifically bladder cancer	X		Section 6.7
Other PFOA and PFOS health effects		X	Section 6.2.2.2
Health effects associated with HI compounds (PFHxS, PFNA, HFPO-DA, PFBS)		X	Section 6.2
Health effects associated with other PFAS		X	Section 6.2

Notes:

1 The national level cost estimates for PFHxS are reflective of both the total national cost for PFHxS individual MCL exceedances, and HI MCL exceedances where PFHxS is present above its HBWC while one or more other HI PFAS is also present in that same mixture. Total quantified national cost values do not include the incremental treatment costs associated with the cooccurrence of HFPO-DA, PFBS, and PFNA. EPA has considered the additional national costs of the HI and individual MCLs associated with HFPO-DA, PFNA, and PFBS occurrence in a quantified sensitivity analysis; see appendix N, section N.3 for the analysis and more information. See appendix N, section N.3 for a sensitivity analysis of additional treatment costs from systems with HI and PFNA and HFPO-DA MCL exceedances. For further discussion of how the EPA considered the costs of the five individual MCLs and the HI MCL, see section XII.A.4 of this preamble.

Sections XII.B to XII.K of this preamble summarize the results of this final rule analysis. The EPA discounted the estimated monetized cost and benefit values using a 2 percent discount rate, consistent with OMB Circular A–4 (OMB, 2003; OMB, 2023) guidance. The U.S. White House and Office of Management and Budget recently finalized and re-issued the A–4 and A–94 benefit-cost analysis guidance (see OMB Circular A–4, 2023), and the update includes new guidance to use a social discount rate of 2 percent. The updated OMB Circular A–4 states that the discount rate should equal the real (inflation-adjusted) rate of return on long-term U.S. government debt, which provides an approximation of the social rate of time preference. This rate for the past 30 years has averaged around 2.0 percent per year in real terms on a pre-tax basis. OMB arrived at the 2 percent discount rate figure by considering the 30-year average of the yield on 10-year Treasury marketable securities, and the approach taken by OMB produces a real rate of 1.7 percent per year, to which OMB added a 0.3 percent per-year rate to reflect inflation as measured by the personal consumption expenditure (PCE) inflation index. The OMB guidance states that Agencies must begin using the 2 percent discount rate for draft final rules that are formally submitted to OIRA after December 31, 2024. The updated OMB Circular A–4 guidance further states that ''to the extent feasible and appropriate, as determined in consultation with OMB, agencies should follow this Circular's guidance earlier than these effective dates.'' Given the updated default social discount rate prescribed in the OMB Circular A–4 and also public input received on the discount rates considered by the EPA in the proposed NPDWR, for this final rule, the EPA estimated national benefits and costs at the 2 percent discount rate for the final rule and incorporated those results into the final economic analysis. Since the EPA proposed this NPDWR with the 3 and 7 percent discount rates based on guidance in the previous version of OMB Circular A–4, the EPA has kept the presentation of results using these discount rates in appendix P. The Administrator reaffirms his determination that the benefits of the rule justify the costs. The EPA's determination is based on its analysis under in SDWA section 1412(b)(3)(C) of the quantifiable benefits and costs at the 2 percent discount rate, in addition to at the 3 and 7 percent discount rate, as well as the nonquantifiable benefits and costs. The EPA found that significant nonquantifiable benefits are likely to occur from the final PFAS NPDWR.

The quantified analysis is limited in its characterization of uncertainty. In section XII.I, Table 68 of this preamble, the EPA provides 5th and 95th percentile values associated with the 2 percent discounted expected values for net benefits. These values represent the quantified, or modeled, potential range in the expected net benefit values associated with the uncertainty resulting from the following variables; the baseline PFAS occurrence; the affected population size; the compliance technology unit cost curves, which are selected as a function of baseline PFAS concentrations and population size, the distribution of feasible treatment technologies, and the three alternative levels of treatment capital costs; the concentration of TOC in a system's source water (which impacts GAC O&M costs); the demographic composition of the system's population; the magnitude of PFAS concentration reductions; the health effect-serum PFOA and PFOS slope factors that quantify the relationship between changes in PFAS serum level and health outcomes for birth weight, CVD, and RCC; and the cap placed on the cumulative RCC risk reductions due to reductions in serum PFOA. These modeled sources of uncertainty are discussed in more detail in section XII.K of this preamble. While the agency reports only the 5th and 95th percentile values, the EPA notes that additional information can be obtained from looking at the whole uncertainty distribution of annualized net benefits (*i.e.,* the distribution of annualized differences between total monetize benefits and total monetized costs).

The quantified 5th and 95th percentile values do not include a number of factors that impact both costs and benefits but for which the agency did not have sufficient data to include in the quantification of uncertainty. The factors influencing the final rule cost estimates that are not quantified in the uncertainty analysis are detailed in Table 43 of this preamble. These uncertainty sources include: the specific design and operating assumptions used in developing treatment unit cost; the use of national average costs that may differ from the geographic distribution of affected systems; the possible future deviation from the compliance technology forecast; and the degree to which actual TOC source water values differ from the EPA's estimated distribution. The EPA has no information to indicate a directional influence of the estimated costs with regard to these uncertainty sources. To the degree that uncertainty exists across the remaining factors it would most likely influence the estimated 5th and 95th percentile range and not significantly impact the expected value estimate of costs.

Table 62 of this preamble discusses the sources of uncertainty affecting the estimated benefits not captured in the estimated 5th and 95th reported values. The modeled values do not capture the uncertainty in: the exposure that results from daily population changes at NTNCWSs or routine population shifting between PWSs, for example spending working hours at a NTNCWS or CWS and home hours at a different CWS; the exposure-response functions used in the benefits analyses assume that the effects of serum PFOA/PFOS on the health outcomes considered are independent, additive, and that there are no threshold serum concentrations below which effects (cardiovascular, developmental, and renal cell carcinoma) do not occur; the distribution of population by size and demographics across EP within modeled systems and future population size and demographic changes; and the Value of Statistical Life reference value or income elasticity used to update the Value of Statistical Life. Given information available to the agency, four of the listed uncertainty sources would not affect the benefits expected value but the dispersion around that estimate. They are the unmodeled movements of populations between PWSs with potentially differing PFAS concentrations; the independence and additivity assumptions with regard to the effects of serum PFOA/PFOS on the health outcomes; the uncertainty in the population and demographic distributions among EP within individual systems; and the Value of Statistical Life value and the income elasticity measures. Two of the areas of uncertainty not captured in the analysis would tend to indicate that the quantified benefits numbers are overestimates. First, the data available to the EPA with regard to population size at NTNCWSs, while likely capturing peaks in populations utilizing the systems, does not account for the variation in use and population and would tend to overestimate the exposed population. The second source of uncertainty, which definitionally would indicate overestimates in the quantified benefits values, is the assumption that there are no threshold serum concentrations below which health effects (cardiovascular, developmental, and renal cell carcinoma) do not occur. One source of possible underestimation of benefits not accounted for in the quantified analysis is the impact of general population growth over the extended period of analysis.

In addition to the quantified cost and benefit expected values, the modeled uncertainty associated within the 5th and 95th percentile values, and the un-modeled uncertainty associated with a number of factors listed above, there are also significant nonquantifiable costs and benefits which are important to the overall weighing of costs and benefits. Table 72 provides a summary of these nonquantifiable cost and benefit categories along with an indication of the directional impact each category would have on total costs and benefits. Tables 43 and 62 also provide additional information on a number of these nonquantifiable categories.

For the nonquantifiable costs, the EPA had insufficient nationally representative data to precisely characterize occurrence of HFPO–DA, PFNA, and PFBS at the national level and therefore could not include complete treatment costs associated with: the co-occurrence of these PFAS at systems already required to treat as a result of estimated PFOA, PFOS, or PFHxS levels, which would shorten the filtration media life and therefore increase operation costs; and the occurrence of HFPO–DA, PFNA, and/or PFBS at levels high enough to cause systems to exceed the individual MCLs for PFNA and HFPO–DA or the Hazard Index and have to install PFAS treatment. The EPA expects that the quantified national costs, which do not include HFPO–DA, PFNA, and PFBS treatment costs are marginally underestimated (on the order of 5%) as a result of this lack of sufficient nationally representative occurrence data. In an effort to better understand and consider the costs associated with treatment of the PFNA and HFPO–DA MCLs and potentially co-occurring HFPO–DA, PFNA, and PFBS at systems both with and without PFOA, PFOS and PFHxS occurrence in exceedance of the MCLs the EPA performed a quantitative sensitivity analysis of the national cost impacts associated with Hazard Index MCL exceedances resulting from HFPO–DA, PFNA, and PFBS and/or individual MCL exceedances of PFNA and HFPO–DA. The analysis is discussed in section 5.3.1.4 and appendix N.3 of the EA (USEPA, 2024l; USEPA, 2024e). Two additional nonquantifiable cost impacts stemming from insufficient co-occurrence data could also potentially shorten filtration media life and increase operation costs. The co-occurrence of other PFAS and other non-PFAS contaminants not regulated in the final rule could both increase costs to the extent that they reduce media life. The EPA did not include POU treatment in the compliance technology forecast because current POU units are not certified to remove PFAS to the standards required in the final rule. Once certified, this technology may be a low-cost treatment alternative for some subset of small systems. Not including POU treatment in this analysis has resulted in a likely overestimate of costs. Additionally, appendix N.2 of the EA (USEPA, 2024e) contains a sensitivity analysis that estimates possible additional national annualized costs of $99 million, which would accrue to systems if the waste filtration media from GAC and IX were handled as RCRA regulatory or characteristic hazardous waste. This sensitivity analysis includes only disposal costs and does not consider other potential environmental benefits and costs associated with the disposal of the waste filtration media.

There are significant nonquantifiable sources of benefits that were not captured in the quantified benefits estimated for the proposed rule. While the EPA was able to monetize some of the PFOA and PFOS benefits related to CVD, infant birth weight, and RCC effects, the agency was unable to quantify additional reductions in negative health impacts in the national quantitative analysis. In addition to the national analysis for the final rule, the agency developed a sensitivity analysis assessing liver cancer impacts, which is detailed in appendix O of the EA (USEPA, 2024e). The EPA did not quantify PFOA and PFOS benefits related to health endpoints including developmental, cardiovascular, hepatic, immune, endocrine, metabolic, reproductive, musculoskeletal, and other types of carcinogenic effects. See section XII.F of this preamble for additional information on the nonquantifiable impacts of PFOA and PFOS. Further, the agency did not quantify any health benefits associated with the potential reductions in Hazard Index PFAS, which include PFHxS, HFPO–DA, PFNA, and PFBS, or other co-occurring non-regulated PFAS which would be removed due to the installation of required filtration technology at those systems that exceed the final MCLs. The nonquantifiable benefits categories associated with exposure to PFHxS, HFPO–DA, PFNA, and PFBS include developmental, cardiovascular, immune, hepatic, endocrine, metabolic, reproductive, musculoskeletal, and carcinogenic effects. In addition, the EPA did not quantify the potential developmental, cardiovascular, immune, hepatic, endocrine, metabolic, reproductive, musculoskeletal, or carcinogenic impacts related to the removal of other co-occurring non-regulated PFAS. See section XII.G of this preamble for additional information on the nonquantifiable impacts of PFHxS, HFPO–DA, PFNA, and PFBS and other non-regulated co-occurring PFAS.

The treatment technologies installed to remove PFAS can also remove numerous other non-PFAS drinking water contaminants which have negative health impacts including additional regulated and unregulated DBPs (the quantified benefits assessment does estimate benefits associated with THM4), heavy metals, organic contaminants, and pesticides, among others. The removal of these co-occurring non-PFAS contaminants could have additional positive health benefits. In total these nonquantifiable benefits are anticipated to be significant and are discussed qualitatively in section 6.2 of the EA (USEPA, 2024g).

To fully weigh the costs and benefits of the action, the agency considered the totality of the monetized values, the potential impacts of the nonquantifiable uncertainties described above, the nonquantifiable costs and benefits, and public comments received by the agency related to the quantified and qualitative assessment of the costs and benefits. For the final rule, the EPA is reaffirming the Administrator's determination made at proposal that the quantified and nonquantifiable benefits of the rule justify its quantified and nonquantifiable costs (88 FR 18638; USEPA, 2023f).

K. Quantified Uncertainties in the Economic Analysis

The EPA characterized sources of uncertainty in its estimates of costs expected to result from the final rule. The EPA conducted Monte-Carlo based uncertainty analysis as part of SafeWater MCBC. With respect to the cost analysis, the EPA modeled the sources of uncertainty in Table 74.

Table 74: Quantified Sources of Uncertainty in Cost Estimates

Source	Description of Uncertainty
EP concentration of PFAS compounds	The concentration and co-occurrence at each PWS EP of each modeled compound is unknown. The cost analysis uses EP concentrations simulated with system level distributions produced by the Bayesian hierarchical Markov chain Monte Carlo (MCMC) occurrence model (see section 4.4 in EA). The iterative MCMC approach (4,000 iterations) probabilistically estimates parameters for system-level distributions to capture uncertainty. The simulated EP concentrations then reflect the system-level distribution from which they are drawn across 4,000 iterations. Further details on the MCMC model are available in Cadwallader et al. (2022). For more information on the application of the model in this analysis, see chapter 4.4 and appendix A. For more information on the data and analyses that the EPA used to develop national estimates of PFAS occurrence in public drinking water systems see USEPA (2024b).
TOC concentration	The TOC value assigned to each system is from a distribution derived from the SYR4 ICR database (see section 5.3.1.1 in EA)
Compliance technology unit cost curve selection	Cost curve selection varies with baseline PFAS concentrations and includes a random selection from a distribution across feasible technologies (see section 5.3.1.2 in EA), and random selection from a triangular distribution of low-, mid-, and high-cost equipment (25 percent, 50 percent, and 25 percent, respectively).

For each iteration, SafeWater MCBC assigned new values to the three sources of modeled uncertainty as described in Table 74, and then calculated costs for each of the model PWSs. This was repeated 4,000 times to reach an effective sample size for each parameter. At the end of the 4,000 iterations, SafeWater MCBC outputs the expected value as well as the 90 percent CI for each cost metric (*i.e.,* bounded by the 5th and 95th percentile estimates for each cost component). Detailed information on the data used to model uncertainty is provided in appendices A and L of USEPA (2024e).

Additionally, the EPA characterized sources of uncertainty in its analysis of potential benefits resulting from changes in PFAS levels in drinking water. The analysis reports uncertainty bounds for benefits estimated in each health endpoint category modeled for the final rule. Each lower (upper) bound value is the 5th (95th) percentile of the category-specific benefits estimate distribution represented by 4,000 Monte Carlo draws.

Table 75 provides an overview of the specific sources of uncertainty that the EPA quantified in the benefits analysis. In addition to these sources of uncertainty, reported uncertainty bounds also reflect the following upstream sources of uncertainty: baseline PFAS occurrence, affected population size and demographic composition, and the magnitude of PFAS concentration reductions. These analysis-specific sources of uncertainty are further described in appendix L of USEPA (2024e).

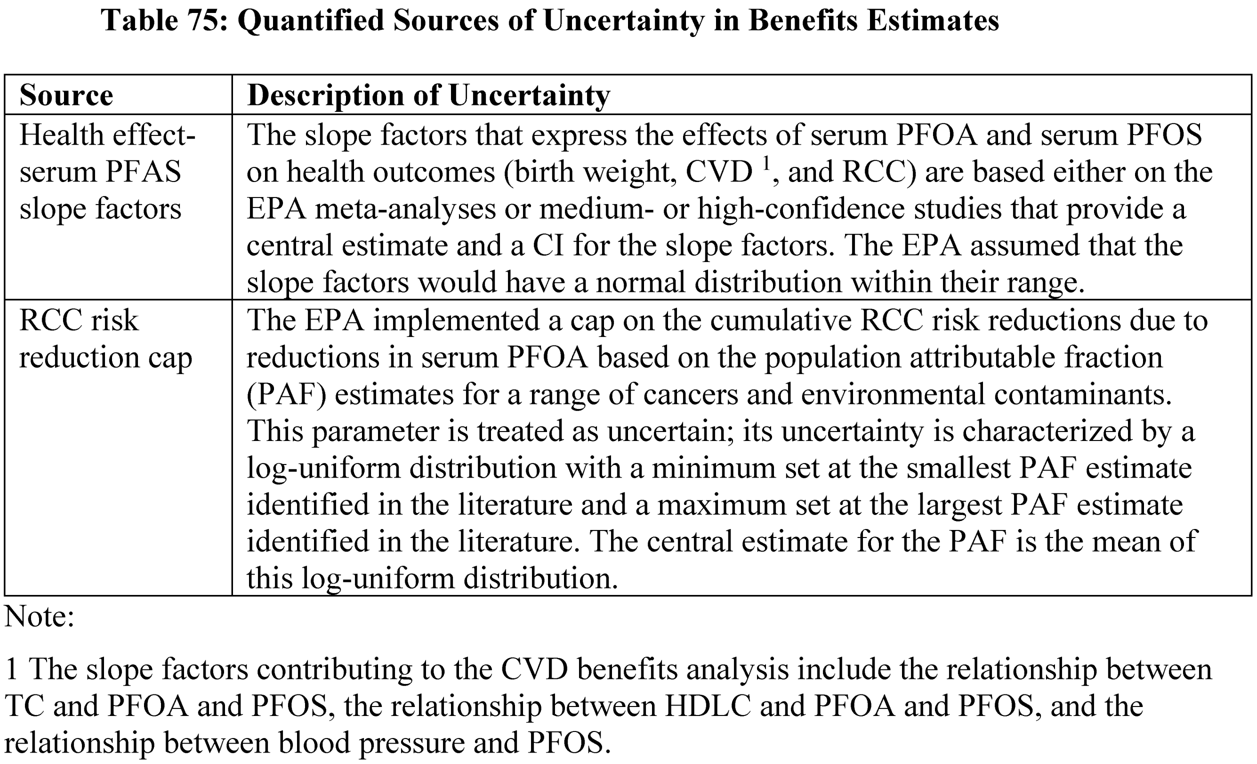

Table 75: Quantified Sources of Uncertainty in Benefits Estimates

Source	Description of Uncertainty
Health effect-serum PFAS slope factors	The slope factors that express the effects of serum PFOA and serum PFOS on health outcomes (birth weight, CVD [1], and RCC) are based either on the EPA meta-analyses or medium- or high-confidence studies that provide a central estimate and a CI for the slope factors. The EPA assumed that the slope factors would have a normal distribution within their range.
RCC risk reduction cap	The EPA implemented a cap on the cumulative RCC risk reductions due to reductions in serum PFOA based on the population attributable fraction (PAF) estimates for a range of cancers and environmental contaminants. This parameter is treated as uncertain; its uncertainty is characterized by a log-uniform distribution with a minimum set at the smallest PAF estimate identified in the literature and a maximum set at the largest PAF estimate identified in the literature. The central estimate for the PAF is the mean of this log-uniform distribution.

Note:

1 The slope factors contributing to the CVD benefits analysis include the relationship between TC and PFOA and PFOS, the relationship between HDLC and PFOA and PFOS, and the relationship between blood pressure and PFOS.

XIII. Statutory and Executive Order Reviews

Additional information about these statutes and Executive orders can be found at *https://www.epa.gov/laws-regulations/laws-and-executive-orders.*

A. Executive Order 12866: Regulatory Planning and Review and Executive Order 14094 Modernizing Regulatory Review

1. Significant Regulatory Action

This action is a ''significant regulatory action,'' as defined under section 3(f)(1) of Executive Order (E.O.) 12866, as amended by E.O. 14094. Accordingly, the EPA submitted this action to the Office of Management and Budget (OMB) for E.O. 12866 review. Documentation of any changes made in response to E.O. 12866 review is available in the docket. The EPA prepared an analysis of the potential costs and benefits associated with this action. This analysis, the Economic Analysis (EA; USEPA, 2024g), is also available in the docket and is summarized in section XII of this preamble.

2. Additional Analysis Under E.O. 12866

The EPA evaluated commenters recommendations summarized in this section to quantify the greenhouse gas (GHG) impacts associated with the rule in light of E.O. 12866, Regulatory Planning and Review, and E.O. 13990, Protecting Public Health and the Environment and Restoring Science to Tackle the Climate Crisis. For the final rule, the EPA has conducted an additional analysis of the disbenefits associated with operation of treatment technologies to comply with the standard. This analysis is summarized here and detailed in the EA for the Final per- and polyfluoroalkyl substances (PFAS) National Primary Drinking Water Regulation (NPDWR; USEPA, 2024g).

a. Proposed Rule

In the proposed rule, the EPA did not quantify and monetize potential GHG emissions impacts that would occur as a result of operating treatment technologies to comply with the proposed rule because quantification of such impacts is not required for the Health Risk Reduction and Cost Analysis (HRRCA) under the Safe Drinking Water Act (SDWA). The EPA evaluated commenters recommendations and summarized that the EPA should quantify and monetize the GHG emissions impacts associated with the rule in light of E.O. 13990, Protecting Public Health and the Environment and Restoring Science to Tackle the Climate Crisis.

b. Summary of Major Public Comments and EPA Responses

Several commenters recommend ''. . . that the agency consider the social costs of carbon as part of any PFAS rule's cost analysis to be comprehensive as well as to understand how this rule may have unintended consequences like increased social costs relating to carbon dioxide emissions.'' Commenters asserted that ''[n]ot including the social costs of carbon and other social costs hinders the Administrator from having all necessary information to set the perfluorooctanoic acid (PFOA) and perfluorooctane sulfonic acid (PFOS) drinking water standard at a level that maximizes health risk reduction benefits at a cost that is justified, given those benefits.'' Commenters pointed to the GHG emissions associated with production, reactivation, and delivery of treatment media, focusing on granular activated carbon (GAC) in particular; construction associated with the installation of the treatment technology at the entry point (EP); electricity used to operate treatment technologies; and transportation and disposal of drinking water treatment residuals to comply with the PFAS NPDWR. Two commenters provided their own quantified estimates for some aspects of CO_2 emissions. One commenter estimated that the climate disbenefits from CO_2 emissions associated with increased electricity use for additional pumping, lighting, and ventilation in treatment plants would be ''$2.5M to $6.8M at 2.5 and 1.5 percent discount rates, respectively, in 2026; and $3.6M to $8.6M at 2.5 and 1.5 percent discount rates, respectively, in 2046.'' Another commenter used a life cycle analysis paper that provides one estimate for the carbon footprint of producing and using GAC and estimates that the climate damages from the CO_2 emissions associated with increased GAC media use ''. . . could have a social cost of more than $160 million annually.'' One commenter stated that the EPA has performed this analysis in other rulemakings, specifically a 2023 proposed air rulemaking (88 FR 25080), and notes that in that regulatory impact analysis (RIA; USEPA, 2023u), ''EPA included the social cost of carbon for the electricity required to operate the air pollution controls.''

The EPA disagrees with commenters that SDWA requires the EPA to quantify and consider the climate disbenefits associated with GHG emission increases from this final rule in the HRRCA. The HRRCA requirements of SDWA 1412 (b)(3)(C) require the agency to analyze ''quantifiable and nonquantifiable costs . . . that are likely to occur *solely* as a result of compliance with the maximum contaminant level'' (emphasis added). Therefore, the EPA considered as part of its HRRCA analysis the compliance costs to facilities, including the costs to purchase electricity required to operate the treatment technologies. Since the climate disbenefits from GHG emissions associated with producing electricity necessary to operate the treatment technologies account for climate impacts associated with the CO_2 emissions and associated costs to society, they do not qualify as compliance costs to public water systems (PWSs) that are part of the required HRRCA analysis under SDWA. For this reason, the EPA included compliance costs to PWSs but not climate disbenefits from GHG emissions associated with the production, reactivation, and delivery of treatment media; construction associated with the installation of the treatment technology at EP; electricity used to operate treatment technologies; and transportation and disposal of drinking water treatment residuals in the cost consideration for the final PFAS NPDWR.

The EPA is committed to understanding and addressing climate change impacts in carrying out the agency's mission of protecting human health and the environment. While the EPA is not required by SDWA 1412(b)(3)(C) to consider climate disbenefits under the HRRCA the agency has estimated the potential climate disbenefits caused by increased on-site electricity demand associated with removing PFAS from drinking water. As explained in section V of this preamble, the EPA's final rule is based on the EPA's record-based analysis of the statutory factors in SDWA 1412(b), and this disbenefits analysis is presented solely for the purpose of complying with E.O. 12866. Circular A–4 states ''[l]ike other benefits and costs, an effort should be made to quantify and monetize additional effects when feasible and appropriate'' (OMB, 2023). The scope of the monetized climate disbenefits analysis is limited to the climate impacts associated with the CO_2 emissions from increased electricity to operate the treatment technologies that will be installed to comply with the PFAS NPDWR.

The EPA did not quantify the potential CO_2 emissions changes associated with the production and delivery of treatment media, construction required for the installation of treatment technology, and transportation and disposal of treatment residuals. The EPA recognizes that many activities directly and indirectly associated with drinking water treatment produce GHG emissions; however, the agency determined that it could not accurately quantify all the potential factors that could increase and decrease greenhouse gas emissions that are not solely attributable to the direct onsite operations of the plant beyond increased electricity use at the plant. The EPA has information, to varying degrees, that the agency could use to potentially estimate emissions from some of these activities. To accurately understand the total potential climate disbenefits of this rule, the EPA should consider GHG emissions in the baseline scenario where the agency also takes no action. However, the EPA lacks the data needed to consider the potentially significant climate disbenefits and other costs to society of the EPA taking no action (*i.e.,* not finalizing the PFAS NPDWR). If the EPA were to not finalize the rule, this could likely trigger other activities that would increase GHG emissions. For example, significant climate disbenefits may be realized from the public increasing purchases of bottled water in an effort to avoid PFAS exposure from drinking water provided by PWSs. More members of the public switch to drinking bottled water if they do not trust the safety of their utility supplied drinking water (Grupper et al. 2021, Levêque and Burns, 2017). Bottled water has a substantially larger carbon footprint than the most highly treated tap water, including the significant energy necessary to produce plastic bottles and transport water from where it is bottled to the point of consumption (Gleick and Cooley, 2009). This carbon footprint can be hundreds of times greater than tap water on a per volume basis (*e.g.,* see Botto, 2009). In addition, this is the first drinking water regulation in which the EPA has estimated disbenefits associated with increases or reductions in GHG emissions. The EPA expects that the approach for quantifying such benefits or disbenefits will continue to evolve as our understanding of the potential relationships between quality of drinking water treatment, impacts on consumer behavior, and other factors influencing GHG emissions improves. Considering the limitations described above and consistent with past EPA rulemakings,[27] the EPA is limiting the scope of the analysis to the major sources of emissions from the direct operation of treatment technologies. The EPA did not quantify the CO_2 emissions associated with production of treatment technologies, construction, transportation, and disposal, as these activities are not solely attributable to the direct onsite operations of the plant and are beyond the scope of this analysis.

Furthermore, while some data exists to inform an estimate of the CO_2 emissions associated with production and reactivation of GAC, the EPA did not do so in this analysis due to significant uncertainties associated with the future CO_2 emissions associated with these technologies. The carbon footprint of GAC is likely to reduce over time, as research continues on novel applications for PFAS removal (*e.g.,* advanced reduction/oxidation processes, novel sorbents, foam fractionation, sonolysis, among others), alternative sources of materials to produce GAC (*e.g.,* biomass and other waste materials), and use of carbon capture technology expands in the future. Given these compounding uncertainties, the EPA did not quantify the climate disbenefits of GAC production and reactivation.

In this rule, the EPA determined that increased electricity use is the major source of emissions from the direct operation of treatment technologies to

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[27] Recent examples include *New Source Performance Standards (NSPS) for the SOC Manufacturing Industry and National Emission Standards for Hazardous Air Pollutants (NESHAP) for the SOC Manufacturing Industry and Group I and Group II polymers and Resins Industry, NESHAP Gasoline Distribution NRPM, Supplemental Effluent Limitations Guidelines (ELGs)* and *Standards for the Steam Electric Power Generating Point Source Category.*

remove PFAS. In this analysis conducted pursuant to E.O. 12866, the EPA first quantified the CO_2 emissions from the additional electricity that is expected to be used for pumping, building lighting, heating, ventilation, and operation of other technology-specific equipment to remove PFAS. The EPA then monetized the climate disbenefits resulting from these CO_2 emissions by applying the social cost of carbon dioxide (SC–CO_2) estimates recommended by the commenter, as described in the following paragraphs.

After considering public comments that recommended the EPA consider the climate disbenefits of the rule, the EPA conducted an analysis similar to the one recommended by one commenter. As suggested by the commenter, the EPA used the estimates of consumption of purchased electricity available from the EPA's peer reviewed work breakdown structure (WBS) cost models to estimate the national electricity use associated with operation of PFAS removal treatment technologies. The EPA deviated from the commenter's suggested approach when estimating associated CO_2 emissions over time from producing electricity. The commenter estimates carbon emissions in a single year and presents that value as a constant reoccurring annual cost. Instead, the EPA estimated how CO_2 emissions would change through 2070, the calendar year to which the EPA has estimated CO_2 emissions from electricity production. The EPA applied readily available information from the latest reference case of the EPA's Integrated Planning Model (IPM) to represent CO_2 emissions associated with electricity production over time.[28] Given that emissions from producing electricity are expected to significantly decrease over time, this is a logical application consistent with other agency rulemakings estimating future emissions from the power sector including the EPA's final *Good Neighbor Plan* (USEPA, 2023q) and the EPA's *New Source Performance Standards for GHG Emissions from New, Modified, and Reconstructed Electric Utility Generating Units* (USEPA, 2023r). Finally, the EPA monetized the climate disbenefits resulting from the estimated CO_2 emissions by applying the SC–CO_2 estimates presented in the regulatory impact analysis of the EPA's December 2023 Final Rule, ''Standards of Performance for New, Reconstructed, and Modified Sources and Emissions Guidelines for Existing Sources: Oil and Natural Gas Sector Climate Review'' (USEPA 2023s). These are the same SC–CO_2 estimates the EPA presented in a sensitivity analysis in the RIA for the agency's December 2022 supplemental proposed Oil and Gas rulemaking that the commenter recommended for use in this action. The SC–CO_2 estimates incorporate recent research addressing recommendations of the National Academies of Science, Engineering, and Medicine (NASEM 2017), responses to public comments on the December 2022 supplemental proposed Oil and Gas rulemaking, and comments from a 2023 external peer review of the accompanying technical report. The methodology underlying the SC–CO_2 estimates is described in the agency's technical report *Report on the Social Cost of Greenhouse Gases: Estimates Incorporating Recent Scientific Advances* (USEPA, 2023t), and is included in the docket for this final rule. For additional details on the climate disbenefits analysis see chapter 9.1 of the EPA's EA for the final PFAS NPDWR.

c. Final Analysis

The EPA did not include an estimate of the monetized climate disbenefits from increased GHG emissions associated with the rule in the HRRCA as recommended by commenters because under the SDWA, the EPA only analyzes compliance costs to PWSs solely as a result of the Maximum Contaminant Level (MCL). The EPA analyzed the climate disbenefits of CO_2 emissions associated with the increased electricity use at PWSs as a result of compliance with the PFAS NPDWR, the EPA estimates annualized climate disbenefits associated with this rule of \$5.5 million per year[29] (under a 2 percent near term discount rate[30]), which constitutes less than 0.4 percent of the monetized benefits of the rule at a 2 percent discount rate. As noted earlier, the EPA's action is justified based on the statutory factors in SDWA section 1412(b) and this disbenefits analysis is presented solely for the purposes of complying with E.O. 12866.

B. Paperwork Reduction Act (PRA)

The information collection activities in this final rule have been submitted for approval to the Office of Management and Budget under the PRA. The Information Collection Request (ICR) document that the EPA prepared has been assigned the EPA ICR number 2732.02 and OMB control number 2040–0307. You can find a copy of the ICR in the docket for this rule at *https://www.regulations.gov/docket/EPA-HQ-OW-2022-0114*, and it is briefly summarized here. The information collection requirements are not enforceable until OMB approves them.

The monitoring information collected as a result of the final rule should allow primacy agencies and the EPA to determine appropriate requirements for specific systems and evaluate compliance with the NPDWR. For the first three-year period following rule promulgation, the major information requirements concern primacy agency activities to implement the rule including adopting the NPDWR into state regulations, providing training to state and PWS employees, updating their monitoring data systems, and reviewing system monitoring data and other requests. Certain compliance actions for drinking water systems, specifically initial monitoring, would be completed during the three years following rule promulgation. Other compliance actions for drinking water systems (including ongoing compliance monitoring, administration, and treatment costs) would not begin until after three years due to the MCL compliance date of this rule. More information on these actions is described in section XII of this preamble and in chapter 9 from the EA of the Final PFAS NPDWR (USEPA, 2024g).

Respondents/affected entities: The respondents/affected entities are PWSs and primacy agencies.

Respondent's obligation to respond: The collection requirements are mandatory under SDWA (42 U.S.C. 300g–7).

Estimated number of respondents: For the first three years after publication of the rule in the **Federal Register**, information requirements apply to an average of 33,594 respondents annually, including 33,538 PWSs and 56 primacy agencies.

Frequency of response: During the initial three-year period, PWSs will conduct one-time startup activities. The one-time burden associated with reading and understanding the rule and adopting the rule is estimated to be an average of 4 hours per system. The one-time burden associated with attending one-time training provided by primacy agencies is an average of 16 hours for systems serving ≤3,300 people and 32 hours for systems serving >3,300 people. The burden associated with initial sampling requirements is an estimated 207,000 hours. The total burden for these activities, for the three-year period, for all systems is estimated to be 1,519,000 hours. During the initial

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[28] See *https://www.epa.gov/power-sector-modeling.*

[29] Disbenefits are annualized over the years 2024–2080.

[30] See the EPA's EA for the Final PFAS NPDWR for results at all discount rates.

three-year period, primacy agencies will incur burdens associated with one-time startup activities. The burden associated with reading and understanding the rule, adopting the regulatory requirements, and training internal staff is estimated to be an average of 4,320 hours per primacy agency. The burden associated with primacy agency review of initial monitoring data is 207,000 hours. The total burden for these activities, for the three-year period, for all 56 primacy agencies is estimated to be 533,000 hours.

Total estimated burden: For the first three years after the final rule is published, water systems and primacy agencies will implement several requirements related to one-time startup activities and monitoring. The total burden hours for public water systems are 1,519,000 hours. The total burden for primacy agencies is 533,000 hours. The total combined burden is 2,052,000 hours.

Total estimated cost: The total costs over the three-year period is $176.8 million, for an average of $58.9million per year (simple average over three years).

An agency may not conduct or sponsor, and a person is not required to respond to, a collected for information unless it displays a currently valid OMB control number. The OMB control numbers for the EPA's regulations in 40 CFR are listed in 40 CFR part 9. When OMB approves this ICR, the agency will announce that approval in the **Federal Register** and publish a technical amendment to 40 CFR part 9 to display the OMB control number for the approved information collection activities contained in this final rule.

C. Regulatory Flexibility Act (RFA)

Pursuant to sections 603 and 609(b) of the RFA, the EPA prepared an initial regulatory flexibility analysis (IRFA) for the proposed rule and convened a Small Business Advocacy Review (SBAR) Panel to obtain advice and recommendations from small entity representatives (SERs) that potentially would be subject to the rule's requirements. Summaries of the IRFA and Panel recommendations are presented in the proposed rule (USEPA, 2023f).

As required by section 604 of the RFA, the EPA prepared a final regulatory flexibility analysis (FRFA) for this action. The FRFA addresses the issues raised by public comments on the IRFA for the proposed rule. The complete FRFA is available for review in section 9.4 of the EA in the docket and is summarized here.

For purposes of assessing the impacts of the final rule on small entities, the EPA considered small entities to be water systems serving 10,000 people or fewer. This is the threshold specified by Congress in the 1996 Amendments to SDWA for small water system flexibility provisions. As required by the RFA, the EPA proposed using this alternative definition in the **Federal Register** (USEPA, 1998d), sought public comment, consulted with the Small Business Administration (SBA), and finalized the small water system threshold in the agency's Consumer Confidence Report (CCR) Regulation (USEPA, 1998e). As stated in the document, the alternative definition would apply to all future drinking water regulations.

The SDWA is the core statute addressing drinking water at the Federal level. Under the SDWA, the EPA sets public health goals and enforceable standards for drinking water quality. As previously described, the final PFAS NPDWR requires water systems to reduce certain PFAS in drinking water below regulatory levels. The EPA is regulating these PFAS in drinking water to improve public health protection by reducing drinking water exposure to these and other PFAS in drinking water.

The final rule contains provisions affecting approximately 62,000 small PWSs. A small PWS serves between 25 and 10,000 people. These water systems include approximately 45,000 community water systems (CWSs) that serve the year-round residents and approximately 17,000 non-transient non-community water systems (NTNCWSs) that serve the same persons over six months per year (*e.g.,* a PWS that is an office or school). The final PFAS NPDWR includes legally enforceable regulatory standards with requirements for monitoring, public notification, and treatment or nontreatment options for water systems exceeding the regulatory standards. This final rule also includes reporting, recordkeeping, and other administrative requirements. States are required to implement operator certification (and recertification) programs under SDWA section 1419 to ensure operators of CWSs and NTNCWSs, including small water system operators, have the appropriate level of certification.

Under the final rule requirements, small CWSs and NTNCWs serving 10,000 or fewer people are required to conduct initial monitoring or demonstrate recent, previously collected monitoring data to determine the level of certain PFAS in their water system. Based on these initial monitoring results, systems are required to conduct ongoing monitoring at least every three years or as often as four times per year. Systems that exceed a drinking water standard will be required to choose between treatment and nontreatment as the compliance option. Under the final rule, the EPA estimates that approximately 16,542 small CWSs (37 percent of small CWSs) could incur annual total PFAS NPDWR related costs of more than one percent of revenues, and that approximately 8,199 small CWSs (18 percent of small CWSs) could incur annual total costs of three percent or greater of revenue. See section 9.3 of the final PFAS NPDWR EA for more information on the characterization of the impacts under the final rule.

The EPA took a number of steps to solicit small entity stakeholder input during the development of the final PFAS NPDWR. Sections XIII.E and XIII.F of this preamble contain detailed information on stakeholder outreach during the rulemaking process, including material on the Federalism and Tribal consultation processes. The EPA also specifically sought input from small entity stakeholders through the SBAR Panel process. On May 24, 2022, the EPA's Small Business Advocacy Chairperson convened the Panel, which consisted of the Chairperson, the Director of the Standards and Risk Management Division within the EPA's Office of Ground Water and Drinking Water, the Administrator of the Office of Information and Regulatory Affairs within OMB, and the Chief Counsel for Advocacy of the SBA. Detailed information on the overall panel process can be found in the panel report available in the PFAS NPDWR docket (EPA–HQ–OW–2022–0114).

In response to the proposal, the EPA received one comment specifically on the analytical approach used in the IRFA. The commenter states that ''[d]etailed analysis on the impacts to NTNCWSs should be conducted to inform the cost/benefit analysis. For example, treating PFAS with GAC at the low levels proposed is much more costly than current treatment for currently regulated contaminants, and a 2008 study is not a reliable indicator of future costs. Lack of both actual data on occurrence in these systems and reliable information on cost of compliance makes finalizing the MCL as to NTNCWSs too uncertain.'' The EPA disagrees that the agency has not analyzed the impacts of the PFAS NPDWR on NTNCWS. The EPA has used both actual data on occurrence at NTNCWSs from the third Unregulated Contaminant Monitoring Rule (UCMR 3) and state data, as well as reliable information on costs to NTNCWSs using the WBS treatment cost models to assess the impact of the rule on NTNCWSs. As the EPA stated in the proposal, the EPA lacks information on the revenues of NTNCWS, therefore the agency does not take the same approach used for CWSs in the Significant Economic Impact on a Substantial Number of Small Entities (SISNOSE) screening analysis where costs are compared to 1 and 3 percent of revenues. Instead, the EPA used the best available data, the EPA's *Assessment of the Vulnerability of Noncommunity Water Systems to SDWA Cost Increases* (USEPA, 1998f), to find that NTNCWSs are less vulnerable to SDWA related increases than a typical CWS. The EPA proceeded with the SBAR Panel process, as previously detailed in this section.

The EPA received many comments on the rule proposal, including from the Chief Counsel for Advocacy of the SBA, on small system and IRFA related topics including lack of funding availability for small water systems, the EPA's alleged underestimation of the impacts of the rule on small systems, the EPA's alleged overestimation of reliance on Federal funding to defray compliance costs for small water systems, and ''other factors that will further deter timely compliance'' such as personnel shortages, supply chain disruptions, limited lab and disposal capacity, and availability of treatment technologies. The EPA has addressed these comments and provided for maximum flexibility for small systems while ensuring sufficient public health protection for populations served by these systems. For the EPA's response to SBA and other comments on funding availability, please see section II of this preamble. For the EPA's response to SBA and other comments on the estimated costs to small water systems, please see section XII of this preamble. For the EPA's response to SBA and other comments on lab capacity, see sections V and VIII. For the EPA's response to SBA and other comments on technology and disposal capacity, see section X. For responses to SBA's and other commenters' recommendations to the EPA to provide burden-reducing flexibilities for small water systems, including finalizing one of the regulatory alternatives and phasing in the MCL, as well as providing additional time for compliance, see section V of this preamble. For response to SBA and other commenters concerned about the EPA's concurrent proposal of a preliminary determination and a proposed regulation for four PFAS, see section III of the preamble. The FRFA, available for review in section 9.4 of the EA in the docket, also provides detailed information on the recommendations of the SBAR Panel and the EPA's actions taken to minimize the significant economic impact of the final rule on small systems.

As a mechanism to reduce the burden of the final rule requirements on small entities the EPA has promulgated compliance flexibilities for small CWSs serving 10,000 or fewer persons. These flexibilities include the use of previously collected PFAS monitoring data to satisfy initial monitoring requirements, allowing reduced initial monitoring for small groundwater systems serving 10,000 or fewer, the addition of annual monitoring to the ongoing compliance monitoring framework, and modified rule trigger levels for reduced monitoring eligibility. For more information on these flexibilities, see section VIII of this preamble. The EPA is also exercising its authority under SDWA section 1412(b)(10) to implement a nationwide two-year capital improvement extension to comply with MCL. The agency notes that SDWA section 1416(a) and (b)(2)(C) describe how the primacy agencies may also grant an exemption for systems meeting specified criteria that provides an additional period for compliance. PWSs that meet the minimum criteria outlined in the SDWA section 1416 may be eligible for an exemption of up to three years. Exemptions for smaller water systems (≤3,300 population), meeting certain specified criteria may be renewed for one or more two-year periods, but not to exceed six years. States exercising primacy enforcement responsibility must have adopted the 1998 Variance and Exemption Regulation for a water system to be eligible for an exemption in that state. Finally, the EPA notes that if point-of-use (POU) devices are certified to meet the NPDWR standard in the future, this could reduce the economic impact of the final regulation on small PWSs, particularly on water systems in the smallest size category (*e.g.,* those serving between 25 and 500 people).

The EPA also assessed the degree to which the final PFAS NPDWR small system flexibilities would mitigate compliance costs. The EPA estimates that the use of previously collected PFAS monitoring data will reduce the economic burden on small systems nationally by $7 million dollars per year for three years. The EPA expects that reduced monitoring for small groundwater systems will reduce the economic burden on small systems nationally by $21 million per year for three years. The EPA estimates that under the final rule approximately 4,300 to 7,000 small PWSs may have regulated PFAS occurrence between the trigger levels and the MCLs, and therefore may be eligible for annual monitoring following four consecutive quarterly samples demonstrating they are ''reliably and consistently'' below the MCLs. The EPA anticipates further compliance cost mitigations stemming from the decision to set the reduced monitoring trigger levels at one-half of the MCLs, rather than one-third of the MCLs as proposed. While the MCL compliance period extension does not change the treatment or non- treatment actions that small systems will be compelled to undertake, it will reduce the compliance burden faced by small water systems by allowing for more time for them to obtain and install capital improvements. Finally, the EPA recognizes the possibility of small system compliance cost reduction particularly for very small water systems should POU certifications be updated in the future and POUs meet the small system compliance technology (SSCT) criteria for the final NPDWR. See chapter 9, section 9.3.4 of the final PFAS NPDWR EA (USEPA, 2024g) for more information on the characterization of the impacts under the final rule.

In addition, the EPA is preparing a Small Entity Compliance Guide to help small entities comply with this rule. The EPA expects the *Small System Compliance Guide* will be developed in the first three years after rule promulgation and will be made available on the EPA's PFAS NPDWR website.

D. Unfunded Mandates Reform Act (UMRA)

This action contains a Federal mandate under UMRA, 2 U.S.C. 1531–1538, that may result in expenditures of $100 million or more for state, local, and Tribal governments, in the aggregate, or the private sector in any one year. Accordingly, the EPA has prepared a written statement required under section 202 of UMRA that is included in the docket for this action (see chapter 9 of the EA for the Final PFAS NPDWR) and briefly summarized here.

Consistent with UMRA section 205, the EPA identified and analyzed a reasonable number of regulatory alternatives to determine the MCL requirement in the final rule. The agency notes, however, that the provisions of section 205 do not apply when they are inconsistent with applicable law; in the case of NPDWRs, the UMRA section 205 requirement to adopt the least costly, most cost-effective, or least burdensome option is inconsistent with SDWA regulatory development requirements. See section XII of this preamble and chapter 9 of the EA for the Final PFAS NPDWR (USEPA, 2024g) for alternative options that were considered. Consistent with the intergovernmental consultation provisions of UMRA section 204, the EPA consulted with governmental entities affected by this rule. The EPA describes the government-to-government dialogue and comments from state, local, and Tribal governments in sections XIII.E. (E.O. 13132: Federalism) and XIII.F. (E.O. 13175: Consultation and Coordination with Indian Tribal Governments) of this document.

This action may significantly or uniquely affect small governments. The EPA consulted with small governments concerning the regulatory requirements that might significantly or uniquely affect them. The EPA describes this consultation in the RFA, section XIII.C. of this preamble.

E. Executive Order 13132: Federalism

The EPA has concluded that this action has federalism implications because it imposes substantial direct compliance costs on state or local governments, and the Federal Government will not provide the funds necessary to pay those costs. However, the EPA notes that the Federal Government will provide a potential source of funds necessary to offset some of those direct compliance costs through the Bipartisan Infrastructure Law (BIL). The EPA estimates that the net change in primacy agency related cost for state, local, and Tribal governments in the aggregate to be $4.7 million.

The EPA provides the following federalism summary impact statement. The EPA consulted with state and local governments early in the process of developing the proposed action to allow them to provide meaningful and timely input into its development. The EPA held a federalism consultation on February 24, 2022. The EPA invited the following national organizations representing state and local elected officials to a virtual meeting on February 24, 2022: The National Governors' Association, the National Conference of State Legislatures, the Council of State Governments, the National League of Cities, the U.S. Conference of Mayors, the National Association of Counties, the International City/County Management Association, the National Association of Towns and Townships, the County Executives of America, and the Environmental Council of States. Additionally, the EPA invited the Association of State Drinking Water Administrators (ASDWA), the Association of Metropolitan Water Agencies (AMWA), the National Rural Water Association (NRWA), the American Water Works Association (AWWA), the American Public Works Association, the Western Governors' Association, the Association of State and Territorial Health Officials, the National Association of Country and City Health Officials, and other organizations to participate in the meeting. In addition to input received during the meeting, the EPA provided an opportunity to receive written input within 60 days after the initial meeting. A summary report of the views expressed during federalism consultations is available in the rule docket (EPA–HQ–OW–2022–0114). The EPA also received public comments from some of these organizations during the public comment period following the rule proposal. These individual organization comments are available in the docket.

Comments provided by the organizations during both the consultation and public comment periods covered a range of topics. The overarching comments from multiple organizations related to the NPDWR compliance timeframe and implementation flexibilities, the proposed MCLs for PFOA and PFOS and the Hazard Index PFAS, the EPA's estimated costs of the NPDWR and funding considerations, PFAS treatment disposal, and other EPA actions to address PFAS in the environment. Specifically, several of these organizations expressed that the EPA should allow an extended compliance timeframe to comply with the MCLs due to supply chain disruptions and availability of treatment materials, as well as maximize the implementation flexibilities for water systems and primacy agencies, including those related to monitoring. Regarding rule costs, some organizations contended that the EPA's costs were underestimated, and that the EPA should consider the disposal of PFAS treatment residuals and associated costs particularly if determined to be hazardous wastes in the future under other EPA statutes such as the Resource Conservation and Recovery Act (RCRA). A couple of organizations requested that the EPA should provide more direct funding for local governments to comply with the NPDWR noting the available BIL funding would not be sufficient to cover the rule costs and these funds cannot be used for certain rule compliance costs. A few organizations suggested that the agency should raise the proposed PFOA and PFOS MCLs, with some of these commenters offering that the EPA should not move forward with the Hazard Index MCL for perfluorohexane sulfonic acid (PFHxS), perfluorononanoic acid (PFNA), hexafluoropropylene oxide dimer acid (HFPO–DA), and perfluorobutane sulfonic acid (PFBS). Finally, several organizations provided that the agency should focus on addressing PFAS holistically and expedite its efforts on source water protection and other actions to address PFAS in the environment beyond drinking water. The EPA considered these organizations' concerns and has taken this input to address many of these in the final PFAS NPDWR while ensuring sufficient public health protection those served by PWSs.

Related to compliance timeline and other rule implementation flexibilities, the EPA is exercising its authority under SDWA section 1412(b)(10) to implement a nationwide two-year capital improvement extension to comply with MCL. The agency notes that SDWA section 1416(a) and (b)(2)(C) describe how the EPA or states may also grant an exemption for systems meeting specified criteria that provides an additional period for compliance. See section XI.D for more information on extensions and exemptions. The EPA has promulgated compliance flexibilities for monitoring implementation including the use of previously collected PFAS monitoring data to satisfy initial monitoring requirements and allowing reduced initial monitoring for small groundwater systems serving 10,000 or fewer. Other monitoring implementation flexibilities include the addition of annual monitoring to the ongoing compliance monitoring framework and higher rule trigger levels for reduced monitoring eligibility. For more information on these flexibilities, see section VIII of this preamble.

For the final rule, the EPA has evaluated the concerns related to the rule costs and maintains that the estimated benefits of the rule justify the costs. Regarding financial costs to water systems if regulated PFAS were to be required to be disposed of as hazardous waste in the future, the EPA reaffirms that no PFAS are currently listed, or proposed to be listed, as hazardous wastes under RCRA. However, the EPA has included a sensitivity analysis to determine the impact on this action should be PFAS-containing treatment materials be considered RCRA regulatory or characteristic hazardous waste in the future (see section X.C. for more detail). For funding concerns and information, the EPA has provided information, detailed further in section II.G. of this preamble related to potential funding opportunities, particularly those available through BIL funds including the EPA's Emerging Contaminants in Small or Disadvantaged Communities (EC–SDC) grants program.

For organizations recommending that the EPA raise the proposed PFOS and PFOS MCLs, with some of these organizations suggesting that the Hazard Index MCL is not justified and should not be finalized, as described in section V of this preamble, the EPA has demonstrated these levels are justified under the requirements of SDWA. Therefore, the agency is maintaining these MCLs for the final rule but has offered compliance flexibilities as described previously.

Lastly, several organizations provided that the agency should focus on addressing PFAS through source water protection efforts beyond drinking water, under the agency's *PFAS Strategic Roadmap* and associated actions, the EPA is swiftly working to address PFAS contamination in the environment and reduce human health PFAS exposure through all pathways. While beyond the scope of this rulemaking, the EPA is making progress implementing many of the commitments in the *Roadmap,* including those that may significantly reduce PFAS source water concentrations.

In addition to the federalism consultation, regarding state engagement more specifically, the EPA notes there were multiple meetings held by ASDWA where the EPA gathered input from state officials and utilized this input to inform this rule. The EPA also considered all comments provided by individual states and state organizations provided during the public comment period and used these comments to inform the final rule.

F. Executive Order 13175: Consultation and Coordination With Indian Tribal Governments

This action has Tribal implications, it imposes direct compliance costs on Tribal governments, and the Federal Government will not provide funds necessary to pay those direct compliance costs. However, the EPA notes that the Federal Government will provide a potential source of funds necessary to offset some of those direct compliance costs through the BIL.

The EPA has identified 998 PWSs serving Tribal communities, 84 of which are federally owned. The EPA estimates that Tribal governments will incur PWS compliance costs of $9.0 million per year attributable to monitoring, treatment or nontreatment actions to reduce PFAS in drinking water, and administrative costs, and that these estimated impacts will not fall evenly across all Tribal systems. The final PFAS NPDWR does offer regulatory relief by providing flexibilities for all water systems to potentially utilize pre-existing monitoring data in lieu of initial monitoring requirements and for groundwater CWSs and NTNCWSs serving 10,000 or fewer to reduce initial monitoring from quarterly monitoring during a consecutive 12-month period to only monitoring twice during a consecutive 12-month period. These flexibilities may result in implementation cost savings for many Tribal systems since 98 percent of Tribal CWSs and 94 percent of NTNCWs serve 10,000 or fewer people.

Accordingly, the EPA provides the following Tribal summary impact statement as required by section 5(b) of E.O. 13175. The EPA consulted with federally recognized Tribal governments early in the process of developing this action to permit them to have meaningful and timely input into its development. The EPA conducted consultation with Indian Tribes beginning on February 7, 2022, and ending on April 16, 2022. The consultation included two national webinars with interested Tribes on February 23, 2022, and March 8, 2022, where the EPA provided proposed rulemaking information and requested input. A total of approximately 35 Tribal representatives participated in the two webinars. Updates on the consultation process were provided to the National Tribal Water Council and the EPA Region 6's Regional Tribal Operations Committee upon request at regularly scheduled monthly meetings during the consultation process. As part of the consultation, the EPA received written comments from the following Tribes: Little Traverse Bay Bands of Odawa Indians and Sault Ste. Marie Tribe of Chippewa Indians. In addition to the comments from these Tribal governments, the EPA received comments the National Tribal Water Council. A summary report of the consultation, webinars, and views expressed during the consultation is available in the Docket (EPA–HQ–OW–2022–0114).

The EPA received a variety of comments from Tribal officials and representatives during both the consultation and public comment periods. These comments can be found in more detail within the Docket through the individual public comments and within the consultation summary report. Specifically, comments included those related to initial monitoring requirements, use of monitoring waivers, concerns related to treatment options and disposal of treatment materials, particularly if determined to be hazardous in the future, as well as funding concerns. The EPA has addressed these officials' comments through finalizing monitoring requirements which allow for small systems flexibilities including the use of previously collected monitoring data to be used to satisfy initial monitoring requirements and not allowing the use of monitoring waivers (see section VIII) of this preamble. Related to treatment considerations, the EPA has identified best available technologies (BATs) as described in section X which have been shown to reduce regulated PFAS levels, but also allows for other treatment technologies not identified as BATs to be used to address MCL exceedances if they can remove PFAS to the regulatory standards. Additionally, the EPA has developed a sensitivity cost analysis to describe the additional financial costs to water systems if the regulated PFAS were to be required to be disposed of as hazardous waste in the future (see appendix N, section 2 of the EA for additional detail). For funding concerns, the EPA has provided information, detailed further in section II of this preamble, related to potential funding opportunities, particularly those available through the EPA's EC–SDC grants program.

The EPA reviewed these comments received from Tribal groups, the estimated cost data, and the quantified and nonquantifiable benefits associated with the PFAS NPDWR and determined that the regulatory burden placed on Tribes is outweighed by the positive benefits. Given that the majority of Tribal systems serve fewer than 10,000 persons, as noted previously, the EPA has provided regulatory relief in the form of small system compliance flexibilities related to monitoring requirements. For additional information on these compliance flexibilities and their estimated impacts see sections VIII of this preamble and chapter 9.4, of the final PFAS NPDWR EA (USEPA, 2024g).

As required by section 7(a) of E.O. 13175, the EPA's Tribal Official has certified that the requirements of the E.O. have been met in a meaningful and timely manner. A copy of the certification is included in the docket for this action.

G. Executive Order 13045: Protection of Children From Environmental Health and Safety Risks

Executive Order 13045 directs Federal agencies to include an evaluation of the health and safety effects of the planned regulation on children in Federal health and safety standards and explain why the regulation is preferable to potentially effective and reasonably feasible alternatives. This action is subject to E.O. 13045 because it is a significant regulatory action under section 3(f)(1) of E.O. 12866, and the EPA believes that the environmental health or safety risk addressed by this action has a disproportionate effect on children. Accordingly, the EPA has evaluated the environmental health or safety effects of the regulated PFAS found in drinking water on children and estimated the risk reduction and health endpoint impacts to children associated with adoption of treatment or nontreatment options to reduce these PFAS in drinking water. The results of these evaluations are contained in the EA of the Final PFAS NPDWR (USEPA, 2024g) and described in section XII of this preamble. Copies of the EA of the Final PFAS NPDWR and supporting information are available in the Docket (EPA–HQ–OW–2022–0114).

Furthermore, the EPA's *Policy on Children's Health* also applies to this action. Information on how the Policy was applied is available in section II.B. of this preamble.

H. Executive Order 13211: Actions That Significantly Affect Energy Supply, Distribution, or Use

This action is not a ''significant energy action'' because it is not likely to have a significant adverse effect on the supply, distribution, or use of energy. The public and private water systems affected by this action do not, as a rule, generate power. This action does not regulate any aspect of energy distribution as the water systems that are proposed to be regulated by this rule already have electrical service. Finally, the EPA has determined that the incremental energy used to implement the identified treatment technologies at drinking water systems in response to the regulatory requirements is minimal. The EPA estimates that the final rule will result in an increased electricity use of approximately 229 GWh per year, for more information see section XIII.A; total U.S. electricity consumption in 2022 was approximately 4.05 million GWh (USEIA, 2023). Therefore, the electricity consumed as a result of the final rule represents approximately 0.005 percent of total U.S. electricity consumption. Based on these findings, the EPA does not anticipate that this rule will have a significant adverse effect on the supply, distribution, or use of energy.

I. National Technology Transfer and Advancement Act of 1995

This action involves technical standards. The rule could involve voluntary consensus standards in that it requires monitoring for regulated PFAS, and analysis of the samples obtained from monitoring based on required methods. As part of complying with this final rule, two analytical methods are required to be used for the identification and quantification of PFAS in drinking water. The EPA Methods 533 and 537.1 incorporate quality control criteria which allow accurate quantitation of PFAS. Additional information about the analytical methods is available in section VII of this preamble. The EPA has made, and will continue to make, these documents generally available through *www.regulations.gov* and at the U.S. Environmental Protection Agency Drinking Water Docket, William Jefferson Clinton West Building, 1301 Constitution Ave. NW, Room 3334, Washington, DC 20460, call (202) 566–2426.

J. Executive Order 12898: Federal Actions To Address Environmental Justice in Minority Populations and Low-Income Populations and Executive Order 14096: Revitalizing Our Nation's Commitment to Environmental Justice for All

1. Proposal

The EPA believes that the human health or environmental conditions that exist prior to this action result in or have the potential to result in disproportionate and adverse human health or environmental effects on communities with environmental justice (EJ) concerns. Consistent with the agency's *Technical Guidance for Assessing Environmental Justice in Regulatory Analysis* (USEPA, 2016f), for the proposed rule, the EPA conducted an EJ analysis to assess the demographic distribution of baseline PFAS drinking water exposure and impacts anticipated to result from the proposed PFAS NPDWR. The EPA conducted two separate analyses: an EJ exposure analysis using the agency's EJSCREENbatch R package, which utilizes data from EJScreen, the agency's Environmental Justice Screening and Mapping Tool (USEPA, 2019e), and from the U.S. Census Bureau's American Community Survey (ACS) 2015–2019 five-year sample (United States Census Bureau, 2022), and an analysis of the EPA's proposed regulatory option and alternatives using SafeWater Multi-Contaminant Benefit Cost Model (MCBC; detailed in section XII of this preamble). The EPA's analyses examined EJ impacts on a subset of PWSs across the country, based on availability of PFAS occurrence data and information on PWS service area boundaries. In the EPA's analysis, results for income, race, and ethnicity groups were generally summarized separately due to how underlying ACS statistics are aggregated at the census block group level; for more information, please see: *https://www.census.gov/data/developers/data-sets/acs-5year.html* (United States Census Bureau, 2022). Additional information on both analyses can be found in chapter 8 of USEPA (2024g) and appendix M of USEPA (2024e).

The EPA's EJ exposure analysis using the EJSCREENbatch R package utilized hypothetical regulatory scenarios, which differed from the EPA's proposed option and regulatory alternatives presented in the proposed rule. The EPA's analysis demonstrated that across hypothetical regulatory scenarios evaluated, elevated baseline PFAS drinking water exposures, and thus greater anticipated reductions in exposure, were estimated to occur in communities of color and/or low-income populations. For this analysis, the EPA examined individuals served by PWSs with modeled PFAS exposure above baseline concentration thresholds or a specific alternative policy threshold. The EPA also summarized population-weighted average concentrations in the baseline as well as reductions that would accrue to each demographic group from hypothetical regulatory scenarios.

The EPA's analysis in SafeWater MCBC evaluated the demographic distribution of health benefits and incremental household costs anticipated to result from the PFAS NPDWR. The EPA's proposed option and all regulatory alternatives were anticipated to provide benefits across all health endpoint categories for all race/ethnicity groups. Across all health endpoints, communities of color were anticipated to experience the greatest reductions in adverse health effects associated with PFAS exposure, resulting in the greatest quantified benefits associated with the EPA's proposed rule, likely due to disproportionate baseline exposure. When examining costs anticipated to result from the rule, the EPA found that cost differences across demographic groups were typically small, with no clear unidirectional trend in cost differences based on demographic group. In some cases, the EPA found that communities of color were anticipated to bear minimally increased costs but in other cases, costs to communities of color were anticipated to be lower than those across all demographic groups. In general, incremental household costs to all race/ethnicity groups were found to decrease with increasing system size, an expected result due to economies of scale.

Additionally, on March 2, 2022, and April 5, 2022, the EPA held public meetings related to EJ and the development of the proposed NPDWR. The meetings provided an opportunity for the EPA to share information and for communities to offer input on EJ considerations related to the development of the proposed rule. During the meetings and in subsequent written comments, the EPA received public comment on topics including establishing an MCL for PFAS, affordability of PFAS abatement options, limiting industrial discharge of PFAS, and the EPA's relationship with community groups. For more information on the public meetings, please refer to the *Environmental Justice Considerations for the Development of the Proposed PFAS Drinking Water Regulation Public Meeting Summary* for each of the meeting dates in the public docket at *https://www.regulations.gov/docket/EPA-HQ-OW-2022-0114*. Additionally, the written public comments are included within the public docket.

2. Summary of Major Public Comments and EPA Responses

Many commenters expressed support for the rule and the EPA's EJ analysis, underscoring the rule's alignment with the administration's commitment to advancing EJ. Commenters point to evidence which suggests that PFAS exposure disproportionately affects communities with EJ concerns. Further, commenters state that these communities are particularly vulnerable to PFAS exposure and the associated health outcomes. Several commenters also assert that the rule is anticipated to benefit these communities with EJ concerns who are at a higher risk of PFAS exposure. Through this rule, the EPA reaffirms the importance of EJ considerations in agency activities, including rulemaking.

Many commenters expressed concern about potential EJ implications of the final rule and urged the EPA to further consider these implications prior to final rule promulgation. Specifically, commenters presented concerns that the rule will disproportionately impact communities that already are overburdened with sociodemographic and environmental stressors. Additionally, several commenters voiced EJ concerns associated with implementation of the rule. Many commenters asserted that communities with EJ concerns may not have sufficient financial capacity to implement the rule (*e.g.,* install treatment) and that this may further exacerbate existing disparities associated with PFAS exposure. Additionally, commenters stated that additional resources would likely be needed for communities with EJ concerns to successfully implement the rule, including targeted monitoring and sampling in these areas.

The EPA acknowledges commenters' concerns regarding potential EJ implications of the rule. Under E.O. 14096, the EPA is directed to identify, analyze, and address disproportionate and adverse human health or environmental effects of agency actions on communities with environmental justice concerns (USEPA, 2023v). The EPA believes that its EJ analysis accompanying the final rule has achieved this directive, as the EPA has assessed the demographic distribution of baseline PFAS exposure in drinking water as well as the anticipated distribution of benefits and costs that will result from the rule. For more information on the EPA's EJ analysis, please see chapter 8 of USEPA (2024g) and appendix M of USEPA (2024e). The EPA acknowledges the potential for implementation challenges for communities with EJ concerns; however, there may be opportunities for many communities to utilize external funding streams to address such challenges. The BIL, the Low-Income Water Household Assistance Program through the American Rescue Plan, and other funding sources may be able to provide financial assistance for addressing emerging contaminants. In particular, the BIL funding has specific allocations for disadvantaged and/or small communities to address emerging contaminants, including PFAS. For example, the *Emerging Contaminants in Small or Disadvantaged Communities (EC–SDC) grants program,* which does not have a cost-sharing requirement, will provide states and territories with $5 billion to provide grants to public water systems in small or disadvantaged communities to address emerging contaminants, including PFAS. Grants will be awarded non-competitively to states and territories.

Many commenters stated that the costs of the rule will disproportionately fall on communities with EJ concerns. Additionally, some commenters asserted that the EPA's EJ analysis does not appropriately consider the distributional impacts of rule costs, with one commenter incorrectly stating that the analysis ''fails to consider how these increased compliance costs will impact EJ communities, as required by Executive Order 12898''. Commenters recommended that the EPA revise its analysis to reflect the impact that compliance costs of the rule will have on communities with potential EJ concerns.

The EPA disagrees with commenters that the EPA has failed to appropriately consider the impact that costs required to implement the rule may have on communities with potential EJ concerns. The agency has fulfilled its commitments in this rulemaking by conducting an analysis consistent with E.O. 14096 and has shared information on the demographic distribution of impacts evaluated in its EJ analysis to facilitate the public's understanding on potential environmental justice impacts of the rule. In section 8.4.2.2 of its EJ Analysis (found in chapter 8 of the HRRCA (USEPA, 2024l)), the EPA estimated the distribution of annualized incremental household costs across different race/ethnicity groups. As described in section XIII.J.1 above, the EPA found that cost differences across demographic groups are typically small, with no clear unidirectional trend in cost differences based on demographic group. In some cases, the EPA found that communities of color are anticipated to bear minimally increased costs but in other cases, costs to communities of color are lower than those across all demographic groups. In response to commenters, the EPA has updated its analysis to also examine the distribution of benefits and costs across income groups. With respect to the distribution of costs, the EPA found that, similar to its findings based on race/ethnicity group, differences in annual incremental household costs across income groups were small with no unidirectional trend in cost differences based on income level.

Additionally, one commenter recommended that the EPA disaggregate Asian and Pacific Islander data in its EJ analysis, asserting that the ''EPA must comply with OMB Statistical Directive 15''. The EPA disagrees that its EJ analysis must disaggregate Asian and Pacific Islander data in order to comply with OMB Statistical Directive 15 (SPD 15). SPD 15 establishes standards for maintaining, collecting, and presenting

Federal data on race and ethnicity and applies to ''all Federal reporting purposes'' (OMB, 1977). This term is not defined and does not clearly apply to analyses developed to support rulemaking efforts. SPD 15 is targeted primarily toward data collection efforts, the development of data for public consumption, and the enforcement of civil rights laws. As SPD 15 is not applicable in the context of rulemakings, the EPA is not required to revise its EJ analysis in accordance with the standards for data disaggregation set forth in the OMB directive. However, the EPA acknowledges that reporting results separately for these groups can help to reveal potential disparities that may exist across Asian and Pacific Islander subpopulations. In response to this comment, the EPA has added a qualitative summary of the literature provided by the commenter and has updated its analysis to include separate Asian and Pacific Islander demographic groups. These updates are reflected in chapter 8 of USEPA (2024g) and appendix M of USEPA (2024e) for the public's information and understanding.

3. Final Rule

The EPA's EJ exposure analysis for the final rule demonstrates that some communities of color are anticipated to experience elevated baseline PFAS drinking water exposures compared to the entire sample population. The percentage of non-Hispanic Black and Hispanic populations with PFAS in drinking water detected above baseline thresholds is greater than the percentage of the total population served with PFAS exposure above these thresholds for all PFAS analytes examined in the EPA's analysis. Similarly, when results are separately analyzed by system size, non-Hispanic Black and Hispanic populations are more likely to be served by large systems with PFAS detected above baseline thresholds compared to the percentage of the total population served across all demographic groups. For small systems, non-Hispanic Asian and non-Hispanic Black populations are more likely to be served by systems with PFAS concentrations above baseline thresholds for some PFAS analytes compared to the total population served across all demographic groups.

The EPA believes that this action is likely to reduce existing disproportionate and adverse effects on communities with EJ concerns. Across all hypothetical regulatory thresholds, elevated exposure—and thus reductions in exposure under the hypothetical regulatory scenarios—is anticipated to occur in communities of color and/or low-income populations. The EPA estimates that the most notable reductions in exposure would be experienced by Hispanic populations, specifically when using UCMR 5 minimum reporting level values as hypothetical regulatory thresholds. Hispanic populations are estimated to experience exposure rates that are at least two percentage points higher than exposure for the total population served across all demographic groups and for all PFAS analytes included in this analysis. Hispanic populations are therefore also expected to have greater reductions in exposure compared to the entire sample population. In addition, under hypothetical regulatory thresholds set at the UCMR 5 minimum reporting levels, the EPA anticipates some of the largest reductions in exposure to PFOA and PFHxS occur for non-Hispanic Native American or Alaska Native and non-Hispanic Pacific Islander populations due to relatively high concentration levels when these PFAS are detected at PWSs serving these groups. For more information on the results of this EJ exposure analysis, see chapter 8 of USEPA (2024g) and appendix M of USEPA (2024e).

For the final rule, the EPA has updated its EJ exposure analysis to include separate Asian and Pacific Islander demographic groups, which were previously combined for the proposed rule. Additionally, the EPA has updated the demographic categories utilized in the EJ exposure analysis to ensure that consistent information is used or applied throughout the PFAS NPDWR EA to the extent possible and to reduce double counting across demographic categories. For the proposed rule, the EPA's EJ exposure analysis used different demographic categories than its distributional analysis conducted in SafeWater, with the former partly including racial groups that were inclusive of Hispanic individuals and the latter including racial groups that were exclusive of Hispanic individuals. Because the EPA's EJ exposure analysis for the proposed rule employed some demographic categories that were inclusive of Hispanic individuals (*e.g.,* American Indian or Alaska Native) and others that were not (*e.g.,* non-Hispanic White), this introduced double counting across groups in the analysis, which complicated making comparisons of exposure across populations of concern. This issue was described in the EJ analysis at proposal, and the EPA solicited comment on alternative methods for defining affected population groups.

Additionally, after considering public comments, the EPA has also updated its EJ analysis conducted in SafeWater MCBC to include an assessment of the distribution of benefits and costs anticipated to result from the final rule across income groups. Findings from the EPA's EJ analysis conducted in SafeWater MCBC for the final rule reaffirm the conclusions of the assessment of the distribution of benefits and costs conducted for the proposed rule across demographic groups. Across all health endpoints evaluated by the EPA, communities of color (*i.e.,* Hispanic, non-Hispanic Black, and/or Other race/ethnicity groups) are anticipated to experience the greatest reductions in adverse health effects associated with PFAS exposure, resulting in the greatest quantified benefits associated with the final rule. For instance, non-Hispanic Black populations are expected to experience 7.48 avoided non-fatal ischemic stroke (IS) cases and 3.90 avoided cardiovascular disease (CVD) deaths per 100,000 people per year, as compared to 3.78 avoided non-fatal IS cases and 1.26 avoided CVD deaths per 100,000 people per year for non-Hispanic White populations. Additionally, under the final rule, while in most cases the difference in cases of illnesses and deaths avoided across income groups is small, quantified health benefits are higher for low-income communities (*i.e.,* populations with income below twice the poverty level) across all health endpoints evaluated, compared to populations with income above twice the poverty level.

As found in its analysis for the rule proposal, when examining costs anticipated to result from the final rule, the EPA found that cost differences across both race/ethnicity and income groups are typically small, with no clear unidirectional trend in cost differences based on demographic group. In some cases, the EPA found that communities of color and low-income communities are anticipated to bear minimally increased costs but in other cases, costs to communities of color and low-income communities are anticipated to be lower than those across all race/ethnicity groups or populations with income above twice the poverty level, respectively. Additionally, incremental household costs to all race/ethnicity and income groups generally decrease as system size increases, which is expected due to economies of scale. This is especially true if systems serving these communities are required to install treatment to comply with the final rule. For example, systems serving 3,300 to 10,000 people that will be required to install treatment to comply with the final rule have substantially higher costs than systems in all larger size categories, irrespective of demographic group. To alleviate potential cost disparities identified by the EPA's analysis, there may be an opportunity for many communities to utilize BIL (Pub. L. 117–58) funding to provide financial assistance for addressing emerging contaminants. BIL funding has specific allocations for both disadvantaged and/or small communities and emerging contaminants, including PFAS.

The information supporting this E.O. 12898 review is contained in chapter 8 of USEPA (2024g) and appendix M of USEPA (2024e) and is available in the public docket for this action. This documentation includes additional detail on the methodology, results, and conclusions of the EPA's EJ analysis.

K. Consultations With the Science Advisory Board, National Drinking Water Advisory Council, and the Secretary of Health and Human Services

In accordance with sections 1412(d) and 1412(e) of the SDWA, the agency consulted with the National Drinking Water Advisory Council (NDWAC, or the Council); the Secretary of U.S. Department of Health and Human Services (HHS); and with the EPA Science Advisory Board (SAB).

1. Science Advisory Board

The SAB PFAS Review Panel met virtually via a video meeting platform on December 16, 2021, and then at three (3) subsequent meetings on January 4, 6, and 7, 2022, to deliberate on the agency's charge questions. Another virtual meeting was held on May 3, 2022, to discuss their draft report. Oral and written public comments were considered throughout the advisory process. The EPA sought guidance from the SAB on how best to consider and interpret life stage information, epidemiological and biomonitoring data, the agency's physiologically based pharmacokinetic (PBPK) analyses, and the totality of PFAS health information to derive an MCLG for PFOA and PFOS, combined toxicity framework, and CVD. The documents sent to SAB were the EPA's *Proposed Approaches to the Derivation of a Draft Maximum Contaminant Level Goal for Perfluorooctanoic Acid (PFOA) (CASRN 335–67–1) in Drinking Water* (USEPA, 2021i); the EPA's *Proposed Approaches to the Derivation of a Draft Maximum Contaminant Level Goal for Perfluorooctane Sulfonic Acid (PFOS) (CASRN 1763–23–1) in Drinking Water* (USEPA, 2021j); the EPA's *Draft Framework for Estimating Noncancer Health Risks Associated with Mixtures of Per- and Polyfluoroalkyl Substances (PFAS)* (USEPA, 2021e); and the EPA's Analysis of Cardiovascular Disease Risk Reduction as a Result of Reduced PFOA and PFOS Exposure in Drinking Water. On May 3 and July 20, 2022, the EPA received input from SAB, summarized in the report *Review of EPA's Analyses to Support EPA's National Primary Drinking Water Rulemaking for PFAS* (USEPA, 2022i).

In response to the EPA's request that the SAB review the EPA's four draft documents listed above, the SAB identified subject matter experts to augment the SAB Chemical Assessment Advisory Committee (CAAC) and assembled the SAB PFAS Review Panel to conduct the review.

In general, the SAB recognized the time constraints for completing the rulemaking process and was supportive of the EPA's efforts to the utilize the latest scientific finding to inform their decisions. The SAB applauded the agency's efforts to develop new approaches for assessing the risk of PFAS mixtures and the benefits arising from reducing exposure to these chemicals as adopted by the EPA in the Hazard Index approach in this rule. In general, the SAB agreed with many of the conclusions presented in the assessments, framework, and analysis. The SAB also identified many areas that would benefit from further clarification to enhance their transparency and increase their utility. The SAB provided numerous recommendations which can be found in the SAB's final report (USEPA, 2022i) and some highlights are outlined in the following section.

a. Approaches to the Derivation of Draft MCLGs for PFOA and PFOS

The primary purpose of the *Proposed Approaches to the Derivation of Draft MCLGs for PFOA and PFOS* (USEPA, 2021i; USEPA, 2021j) was to develop Maximum Contaminant Level Goals (MCLGs) based on the best available health effects information for PFOA and PFOS. Each MCLG draft document includes derivation of an updated chronic oral reference dose (RfD), cancer slope factor (CSF) when relevant data were available, and a relative source contribution (RSC) for SAB review. The health effects information used to derive these toxicity values and RSC values built upon the information in the 2016 EPA PFOA and PFOS Health Effects Support Documents (HESDs; USEPA, 2016c; USEPA, 2016d). The EPA has considered all SAB consensus advice in the development of the final values derived in this health effects assessment and subsequently derived MCLGs for the NPDWRs for PFOA and PFOS based on the best available science and the EPA guidance and precedent. Please see section IV of this preamble for discussions on the process for derivation of the MCLGs and the resulting proposed MCLG values for this final action.

The SAB charge questions for the MCLG draft documents addressed the systematic review study identification and inclusion, non-cancer hazard identification, cancer hazard identification and slope factor, toxicokinetic (TK) modeling, RfD derivation, and RSC. The complete list of charge questions was included in the EPA's documents prepared for the SAB (USEPA, 2022i). The SAB provided numerous specific recommendations to consider alternative approaches, expand the systematic review steps for the health effects assessment, and to develop additional analyses in order to improve the rigor and transparency of the EPA's documents. The complete list of SAB consensus advice is described in their final report (USEPA, 2022i).

Regarding the approaches to deriving MCLG draft documents, the SAB stated that the systematic review methods could be more transparent and complete. Specifically, study identification and criteria for inclusion could be improved. The EPA made revisions to the systematic review description and process by updating and expanding the scope of the literature search; providing greater transparency regarding the study inclusion criteria; and adding additional systematic review steps and transparently describing each of these steps in the PFOA and PFOS systematic review protocols.

In the charge questions, the EPA sought advice on the noncancer health assessment, and the SAB recommended that the EPA separate hazard and dose-response assessment systematic review steps. In response, the EPA made revisions to the noncancer hazard identification by expanding systematic review steps beyond study quality evaluation to include evidence integration to address the need to separate hazard identification and dose-response assessment and to ensure consistent hazard decisions; and strengthening rationales for selection of points of departure for the noncancer health outcomes. Additionally, the SAB advised the EPA to focus on the health endpoints with the strongest evidence (*i.e.,* liver, immune, serum lipids, development, and cancer).

The EPA consulted with the SAB on the cancer risk assessment. On the cancer Hazard Index and CSF, the SAB agreed that PFOA was a ''likely'' designation but recommended undertaking and describing a more structured and transparent discussion of the ''weight of evidence'' for both PFOA and PFOS. The EPA revised this assessment by following the structured approach in the EPA cancer guidelines (USEPA, 2005a) to develop a weight of evidence narrative for cancer, to consider the data for selecting the cancer classification, evaluating and integrating mechanistic information, and strengthening the rationales for decisions.

With respect to the TK model for which the EPA sought advice, SAB requested more details on the TK modeling including model code and parameters and recommended that the EPA consider expressing the RfD in water concentration equivalents to better account for possible life-stage specific differences in exposure rates and TKs. The EPA considered the alternate approach suggested by SAB and made revisions by evaluating alternative TK models and further validating the selected model.

The EPA also sought advice on the draft RfD derivation. The SAB advised that the EPA consider multiple human and animal studies for a variety of endpoints and populations. The SAB also stated a need for stronger and more transparent justification of BMR selections and asked the EPA to consider adopting a probabilistic framework to calculate risk-specific doses. SAB also recommended that the EPA clearly state that RfDs apply to both short-term and chronic exposure. The EPA made revisions based on these recommendations by providing additional descriptions and rationale for the selected modeling approaches and conducting new dose-response analyses of additional studies and endpoints.

On the RSC charge question, SAB supported the selection of a 20 percent RSC, but asked that the EPA provide clarity and rationale to support the value. To address this recommendation, the EPA added clarifying language related to the RSC determination from the EPA guidance (USEPA, 2000d), including the relevance of drinking water exposures and the relationship between the RfD and the RSC.

b. Combined Toxicity Framework

The EPA sought advice from the SAB on the *Draft Framework for Estimating Noncancer Health Risks Associated with Mixtures of PFAS* document (USEPA, 2021e). The main purpose of this document was to provide a data-driven framework for estimating human health risks associated with oral exposures to mixtures of PFAS. The charge questions for the SAB pertaining to the framework draft documents included whether the EPA provided clear support for the assumption of dose additivity, and application of the Hazard Index, relative potency factor (RPF), and mixtures benchmark dose (BMD) approaches for the evaluation of mixtures of PFAS. The full list of charge questions was included in the EPA's documents prepared for the SAB (USEPA, 2022i). The SAB agreed in general with dose additivity at the level of common health effect, and application of the Hazard Index, RPF and mixture BMD approaches for the evaluation of mixtures of PFAS. The SAB identified instances in which the communication of the analyses and approaches in the EPA's framework document could be improved to be clearer.

On the EPA's charge question for dose additivity, the SAB agreed with the use of the dose additivity assumption when evaluating PFAS mixtures that have similar effects and concluded that this approach was health protective. The SAB recommended a more thoroughly and clearly presented list of the uncertainties associated with dose additivity along with information supporting this approach. The EPA made revisions that added clarity to the text by expanding upon the uncertainties and including additional support for using dose additivity.

The SAB panel agreed with the use of the Hazard Index as a screening method and decision-making tool. The SAB advised that the EPA should consider using a menu-based framework to support selection of fit-for-purpose approaches, rather than a tiered approach as described in the draft mixtures document. Based on this feedback, the EPA has since reorganized the approach to provide a data-driven ''menu of options'' to remove the tiered logic flow and is adding text to clarify the flexibility in implementation.

The EPA sought the SAB's opinion on the RPF approach for estimating health risks associated with PFAS mixtures and the SAB panel considered the RPF approach to be a reasonable methodology for assessing mixtures. On the mixture BMD, the SAB agreed that the mixture BMD approach was a reasonable methodology for estimating a mixture-based point of departure (POD). For both the RPF and mixture BMD approach, the SAB recommended that the EPA's approach be strengthened by the use of PODs from animal studies that are based on HEDs rather than administered doses. The SAB also requested clarification as to the similarities and differences among the RPF and mixture BMD approaches. The SAB also asked the EPA to provide additional information on how the proposed mixtures BMD approach would be applied in practice. To address these recommendations, the EPA made revisions to provide better context and delineation about the applicability of the data across these approaches.

c. Cardiovascular Disease Analysis

The EPA consulted with the SAB on the agency's methodology to determine the avoided cases of CVD events (*e.g.,* heart attack, stroke, death from coronary heart disease) associated with reductions in exposure to PFOA and PFOS in drinking water to support a benefits analysis. Specifically, the EPA sought SAB comment on the extent to which the approach to estimating reductions in CVD risk is scientifically supported and clearly described. The EPA posed specific charge questions on the exposure-response information used in the analysis, the risk model and approach used to estimate the avoided cases of CVD events, and the EPA's discussion of limitations and uncertainties of the analysis. Overall, the SAB supported the EPA's approach to estimating reductions in CVD risk associated with reductions in exposure to PFOA and PFOS in drinking water. The SAB provided feedback on several areas of the analysis; main points of their feedback and the EPA's responses are discussed in this section.

The SAB noted a discrepancy between the draft CVD document's focus on CVD risk, and the draft MCLG documents' conclusions that the evidence of CVD was not sufficient to form the basis of a RfD. Based on SAB feedback on the draft MCLG document's assessment of CVD related risks, the EPA has developed an RfD for total cholesterol (TC). (For more information see USEPA, 2024c; USEPA, 2024d.) The derivation of an RfD for this endpoint addresses the SAB's concerns about inconsistency between the two documents. The SAB also recommended that the EPA ensure that recommendations for the draft MCLG documents relating to evidence identification and synthesis are applied to the CVD endpoint. All studies in the EPA's CVD benefits analysis were evaluated for risk of bias, selective reporting, and sensitivity as applied in the EPA's *Public Comment Draft—Toxicity Assessment and Proposed MCLGs for PFOA and PFOS in Drinking Water* (USEPA, 2023g; USEPA, 2023h).

The SAB recommended that the EPA provide more discussion as to the rationale for selecting CVD for risk reduction analysis and that the approach follows the pathway that links cholesterol to cardiovascular events rather than looking at the reported effects of PFAS directly on CVD. The SAB also recommended that the EPA consider risk reduction analyses for other endpoints. In section 6.5 of the EA, the EPA discusses the rationale for quantifying CVD and analytical assumptions. Sections 6.4 and 6.6 discusses the agency's quantified risk reduction analyses for other adverse health effects, including infant birthweight effects and renal cell carcinoma (RCC), respectively. In section 6.2.2, the EPA assesses the qualitative benefits of other adverse health effects of PFAS.

Although the SAB generally agreed with the meta-analysis, life table and risk estimation methods, the SAB recommended that the EPA provide additional clarity as to the application of these approaches and conduct additional sensitivity analyses. In response to these comments, the EPA expanded documentation and conducted additional sensitivity analyses to evaluate the impact of inclusion or exclusion of certain studies in the meta-analyses of exposure-response estimates. Further, the EPA expanded documentation and conducted additional sensitivity analyses to assess the effects of using a key single study approach versus the meta-analysis approach to inform the exposure-response estimates. The EPA identified two suitable key studies for use in the single study approach. The EPA found that the single study approach resulted in increased benefits, and this trend was driven by the larger estimates of PFAS–TC slope factors and inverse associations in the high-density lipoprotein cholesterol (HDLC) effect for one or both contaminants in the key single studies. The EPA elected to retain the meta-analysis approach in the benefits analysis because the agency identified several studies on adults in the general population with large numbers of participants and low risk of bias, and in this case the meta-analytical approach offers an increased statistical power over the single study approach. While the single study approach is common for RfD derivations, the meta-analysis pooled estimate provides a slope factor that represents the average response across a larger number of studies, which is useful in evaluating benefits resulting from changes in CVD risk on a national scale.

The SAB also recommended that the EPA evaluate how inclusion of HDLC effects would influence the results and provide further justification for the inclusion or exclusion of HDLC and blood pressure effects. The EPA found that, as expected, inclusion of HDLC effects decreases annualized CVD benefits and inclusion of blood pressure effects slightly increases annualized CVD benefits. Because HDLC was shown to have a stronger effect than blood pressure on annualized CVD benefits, inclusion of blood pressure and HDLC effects together decreases annualized CVD benefits. For more information see sensitivity analyses evaluating these effects in appendix K of the EA. Inclusion of HDLC effects into the national analysis would reduce national benefits estimates but would not change the EPA's bottom-line conclusion that the quantifiable and nonquantifiable benefits of the rule justify the quantifiable and nonquantifiable costs. After further examination of the evidence for HDLC and blood pressure effects, the EPA elected to include blood pressure effects because the findings from a single high confidence study and several medium confidence studies conducted among the general population provided consistent evidence of an association between PFOS exposure and blood pressure. The EPA did not include HDLC effects in the national benefits analysis because available evidence of associations between PFOS exposures and HDLC levels is inconsistent and there is no evidence of an association between PFOA exposures and HDLC levels.

Finally, the SAB noted that while the Atherosclerotic Cardiovascular Disease (ASCVD) model is a reasonable choice for estimating the probability of first time CVD events, it is not without limitations. The panel recommended that the EPA include more discussion of the accuracy of its predictions, particularly for sub-populations. The EPA expanded its evaluation of the ASCVD model's limitations, including a comparison of the ASCVD model predictions with race/ethnicity and sex-specific CVD incidence from Centers for Disease Control and Prevention's (CDC's) public health surveys (See section 6.5.3.2 and appendix G of the EA for details). Results show that the ASCVD model coefficients for the non-Hispanic Black model are more consistent with data on CVD prevalence and mortality for Hispanic and non-Hispanic other race subpopulations than the ASCVD model coefficients for the non-Hispanic White model.

Comments on the SAB consultation and review were raised by public commenters. As a result, the comments have been addressed by the EPA in the final rule, supporting documents in the record, and throughout this preamble, specifically in sections III.B, IV, and XII.A.

2. National Drinking Water Advisory Council (NDWAC)

The agency consulted with the NDWAC prior to the rule proposal during the Council's April 19, 2022, virtual meeting. During the meeting, the EPA provided information related to the development of the proposed rule. A summary of the NDWAC input from that meeting is available in the *NDWAC, Fall 2022 Meeting Summary Report* (NDWAC, 2022) and the docket.

On August 8, 2023, the EPA consulted with the NDWAC prior to the final rule during a virtual meeting where the EPA presented on the proposed PFAS NPDWR, including the proposed MCLs, monitoring and PN requirements, and treatment and economic considerations. The EPA reiterated that the PFAS NPDWR was developed with extensive consultation from state, local and Tribal partners to identify avenues that would reduce PFAS in drinking water and reaffirmed its commitment to working with these partners on rule implementation. The EPA carefully considered the information provided by the NDWAC during the development of a final PFAS NPDWR. A summary of the NDWAC input from that meeting is available in the *NDWAC Summary Report* (NDWAC, 2023) and the docket.

3. Department of Health and Human Service

On September 28, 2022, the EPA consulted with the Department of HHS on the proposed PFAS NPDWR. On November 2, 2023, the EPA consulted with the HHS on the final rule. The EPA received and considered comments from the HHS for both the proposed and final rules through the interagency review process described in section XIII.A.

L. Congressional Review Act (CRA)

This action is subject to the CRA, and the EPA will submit a rule report to each House of the Congress and to the Comptroller General of the United States. This action meets the criteria set forth in 5 U.S.C.804(2).

XIV. Severability

The purpose of this section is to clarify the EPA's intent with respect to the severability of provisions of this rule. Each Maximum Contaminant Level (MCL) is independent of the others and can be implemented on its own. For that reason, if any individual or Hazard Index MCL is determined by judicial review or operation of law to be invalid, the EPA intends that the partial invalidation will not render any other

MCL invalid. In addition, each per- and polyfluoroalkyl substance (PFAS) included in the Hazard Index is independent from any other PFAS included in the Hazard Index. As a result, if any PFAS regulation is determined by judicial review or operation of law to be invalid, that partial invalidation should not render any other PFAS regulation included in the Hazard Index or the Hazard Index PFAS MCL invalid. Moreover, the Hazard Index approach and Hazard Index PFAS MCL can remain operable and applicable so long as there are at least two contaminants subject to the Hazard Index as a mixture because the EPA's definition of mixture in this final rule is of two or more of the Hazard Index PFAS. In addition, each individual Maximum Contaminant Level Goal (MCLG) is independent of each of the other MCLGs and, because they perform different functions under the Act, of each of the MCLs. As a result, if an MCL is determined to be invalid, that partial invalidation should not render the associated MCLG invalid. The monitoring requirements are independent and capable of operating without any MCLs. Likewise, if any provision of this rule other than the MCLGs, or MCLs, is determined to be invalid (such as monitoring waivers or the capital improvements extension), the remainder of the rule can still be sensibly implemented; as a result, the EPA intends that the rest of the rule (such as monitoring requirements) remain operable and applicable.

XV. Incorporation by Reference

In this action, the EPA requires that systems must only use the analytical methods specified to demonstrate compliance with the rule. EPA Method 533: Determination of Per- and Polyfluoroalkyl Substances in Drinking Water by Isotope Dilution Anion Exchange Solid Phase Extraction and Liquid Chromatography/Tandem Mass Spectrometry, November 2019, 815–B–19–020, and EPA Method 537.1,Version 2.0: Determination of Selected Per- and Polyfluorinated Alkyl Substances in Drinking Water by Solid Phase Extraction and Liquid Chromatography/Tandem Mass Spectrometry (LC/MS/MS), March 2020, EPA/600/R–20/006, are incorporated by reference in this final rule and are publicly available in the EPA's Docket ID No. EPA–HQ–OW–2022–0114. The EPA Method 533 and EPA Method 537.1, Version 2.0 are solid phase extraction liquid chromatography-tandem mass spectrometry methods for the detection and determination of select per-and polyfluoroalkyl substances in drinking water. In addition to being available in the aforementioned rule docket, both methods can be accessed online at *https://www.epa.gov/pfas/epa-pfas-drinking-water-laboratory-methods*.

XVI. References

3M (3M Company). 2000. Sanitized report: Exploratory 28-day oral toxicity study with telomer alcohol, telomer acrylate, PFHS and PFOS (positive control) by daily gavage in the rat followed by a 14/28-day recovery period [TSCA Submission]. (FYI–0500–01378 B; DCN 84000000018). St. Paul, MN. *https://yosemite.epa.gov/oppts/epatscat8.nsf/ReportSearchView/05ADDF3067814C978525703B004B4119*.

Adamson, D.T., Piña, E.A., Cartwright, A.E., Rauch, S.R., Anderson, R.H., Mohr, T., and Connor, J.A. 2017. 1,4-Dioxane drinking water occurrence data from the third unregulated contaminant monitoring rule. *Science of the Total Environment,* 596:236–245.

Aguilera, P.A., Fernández, A., Fernández, R., Rumí, R., and Salmerón, A. 2011. Bayesian networks in environmental modelling. *Environmental Modelling & Software,* 26(12):1376–1388.

Almond, D., Chay, K.Y., and Lee, D.S. 2005. The Costs of Low Birth Weight. *The Quarterly Journal of Economics,* 120(3):1031–1083. *https://doi.org/10.1093/qje/120.3.1031*.

Ambavane, A., Yang, S., Atkins, M.B., Rao, S., Shah, A., Regan, M.M., McDermott, D.F., and Michaelson, M.D. 2020. Clinical and Economic Outcomes of Treatment Sequences for Intermediate- to Poor-Risk Advanced Renal Cell Carcinoma. *Immunotherapy,* 12(1):37–51. *https://doi.org/10.2217/imt-2019-0199*.

Agency for Toxic Substances and Disease Registry (ATSDR). 2021. *Toxicological Profile for Perfluoroalkyls.* U.S. Department of Health and Human Services, Agency for Toxic Substances and Disease Registry, Atlanta, GA. Accessed February 2022. *https://www.atsdr.cdc.gov/toxprofiles/tp200.pdf*.

American Cancer Society. 2020. Cancer facts and figures 2019 key statistics about kidney cancer. Available at: *https://hero.epa.gov/hero/index.cfm/reference/details/reference_id/9642148*.

American Water Works Association (AWWA). 2023. AWWA Comments on the Proposed ''PFAS National Primary Drinking Water Regulation Rulemaking.'' Available at: *https://www.regulations.gov/comment/EPA-HQ-OW-2022-0114-1465*.

Appleman, T.D., Higgins, C.P., Quiñones, O., Vanderford, B.J., Kolstad, C., Zeigler-Holady, J.C., and Dickenson, E.R. 2014. Treatment of poly- and perfluoroalkyl substances in US full-scale water treatment systems. *Water Research,* 51:246–255. *https://doi.org/10.1016/j.watres.2013.10.067*.

Averina, M., Brox, J., Huber, S., and Furberg, A.S. 2021. Exposure to perfluoroalkyl substances (PFAS) and dyslipidemia, hypertension and obesity in adolescents. The Fit Futures study. *Environmental Research,* 195:110740. *https://doi.org/10.1016/j.envres.2021.110740*.

Bach, C.C., Bech, B.H., Nohr, E.A., Olsen, J., Matthiesen, N.B., Bonefeld-Jørgensen, E.C., Bossi, R., and Henriksen, T.B. 2016. Perfluoroalkyl acids in maternal serum and indices of fetal growth: The Aarhus Birth Cohort. *Environmental Health Perspectives,* 124:848–854. *http://dx.doi.org/10.1289/ehp.1510046*.

Bach, C.C., Matthiesen, N.B., Olsen, J., and Henriksen, T.B. 2018. Conditioning on parity in studies of perfluoroalkyl acids and time to pregnancy: an example from the Danish National Birth Cohort. *Environmental Health Perspectives,* 126(11):117003.

Bao, W.W., Qian, Z.M., Geiger, S.D., Liu, E., Wang, S.Q., Lawrence, W.R., Yang, B.Y., Hu, L.W., Zeng, X.W., and Dong, G.H. 2017. Gender-specific associations between serum isomers of perfluoroalkyl substances and blood pressure among Chinese: isomers of C8 health project in China. *Science of the Total Environment,* 607–608:1304–1312. doi:10.1016/j.scitotenv.2017.07.124.

Barry, V., Winquist, A., and Steenland, K. 2013. Perfluorooctanoic Acid (PFOA) Exposures and Incident Cancers among Adults Living Near a Chemical Plant. *Environmental Health Perspectives,* 121(11–12):1313–1318. *https://doi.org/10.1289/ehp.1306615*.

Bartell, S.M., Calafat, A.M., Lyu, C., Kato, K., Ryan, P.B., and Steenland, K. 2010. Rate of Decline in Serum PFOA Concentrations after Granular Activated Carbon Filtration at Two Public Water Systems in Ohio and West Virginia. *Environmental Health Perspectives,* 118(2):222–228. *https://doi.org/10.1289/ehp.0901252*.

Bartell, S.M., and Vieira, V.M. 2021. Critical Review on PFOA, Kidney Cancer, and Testicular Cancer. *Journal of the Air & Waste Management Association,* 71(6):663–679. *https://doi.org/10.1080/10962247.2021.1909668*.

Beane Freeman, L.E., Cantor, K.P., Baris, D., Nuckols, J.R., Johnson, A., Colt, J.S., Schwenn, M., Ward, M.H., Lubin, J.H., Waddell, R., Hosain, G.M., Paulu, C., McCoy, R., Moore, L.E., Huang, A., Rothman, N., Karagas, M.R., and Silverman, D.T. 2017. Bladder Cancer and Water Disinfection By-product Exposures through Multiple Routes: A Population-Based Case-Control Study (New England, USA). *Environmental Health Perspectives,* 125(6). *https://doi.org/10.1289/EHP89*.

Behrman, J.R., and Rosenzweig, M.R. 2004. Returns to Birthweight. *The Review of Economics and Statistics,* 86(2):586–601. *https://doi.org/10.1162/003465304323031139*.

Behrman, R.E., and Butler, A.S. 2007. Preterm Birth: Causes, Consequences, and Prevention. Institute of Medicine (US) Committee on Understanding Premature Birth and Assuring Healthy Outcomes. National Academies Press 2007. *https://doi.org/10.17226/11622*.

Bellia, G.R., Bilott, R.A., Sun, N., Thompson, D., and Vasiliou, V. 2023. Use of clinical chemistry health outcomes and PFAS chain length to predict 28-day rodent oral toxicity. *Toxicology Mechanisms and Methods,* 33(5):378–387.

Berretta, C., Mallmann, T., Trewitz, K., and Kempisty, D.M. 2021. Removing PFAS from Water: From Start to Finish. In Kempisty, D.M. and Racz, L. (Eds.), Forever Chemicals: Environmental, Economic, and Social Equity Concerns with PFAS in the Environment (pp. 235–253). CRC Press. *https://doi.org/10.1201/9781003024521.*

Blake, B.E., Pinney, S.M., Hines, E.P., Fenton, SE, and Ferguson, K.K. 2018. Associations between longitudinal serum perfluoroalkyl substance (PFAS) levels and measures of thyroid hormone, kidney function, and body mass index in the Fernald Community Cohort. *Environmental Pollution,* 242:894–904.

Bonefeld-Jorgensen, E.C., Long, M., Fredslund, S.O., Bossi, R., and Olsen, J. 2014. Breast cancer risk after exposure to perfluorinated compounds in Danish women: A case-control study nested in the Danish National Birth Cohort. *Cancer Causes Control,* 25(11):1439–1448.

Boobis, A., Budinsky, R., Collie, S., Crofton, K., Embry, M., Felter, S., Hertzberg, R., Kopp, D., Mihlan, G., Mumtaz, M., Price, P., Solomon, K., Teuschler, L., Yang, R., and Zaleski, R. 2011. Critical analysis of literature on low-dose synergy for use in screening chemical mixtures for risk assessment. *Critical Reviews in Toxicology,* 41(5):369–383. *https://doi.org/10.3109/10408444.2010.543655.*

Boone, L., Meyer, D., Cusick, P., Ennulat, D., Bolliger, A.P., Everds, N., . . . Bounous, D. 2005. Selection and interpretation of clinical pathology indicators of hepatic injury in preclinical studies. *Veterinary Clinical Pathology,* 34(3):182–188.

Botto, S. 2009. Tap Water vs. Bottled Water in a Footprint Integrated Approach. *Nature Proceedings, https://doi.org/10.1038/npre.2009.3407.1.*

Budtz-Jørgensen, E., and Grandjean, P. 2018. Application of Benchmark Analysis for Mixed Contaminant Exposures: Mutual Adjustment of Perfluoroalkylate Substances Associated with Immunotoxicity. *PLoS ONE,* 13:e0205388. *https://doi.org/10.1371/journal.pone.0205388.*

Burkhardt, J.B., Cadwallader, A., Pressman, J.G., Magnuson, M.L., Williams, A.J., Sinclair, G., and Speth, T.F. 2023. Polanyi adsorption potential theory for estimating PFAS treatment with granular activated carbon. *Journal of Water Process Engineering,* 53:103691.

Butenhoff, J.L., Chang, S.C., Ehresman, D.J., and York, R.G. 2009. Evaluation of potential reproductive and developmental toxicity of potassium perfluorohexanesulfonate in Sprague Dawley rats. *Reproductive Toxicology,* 27(3–4):331–341.

Butenhoff, J.L., Chang, S.C., Olsen, G.W., and Thomford, P.J. 2012. Chronic Dietary Toxicity and Carcinogenicity Study with Potassium Perfluorooctane Sulfonate in Sprague Dawley Rats. *Toxicology,* 293:1–15. *https://doi.org/10.1016/j.tox.2012.01.003.*

Cadwallader, A., Greene, A., Holsinger, H., Lan, A., Messner, M., Simic, M., and Albert, R. 2022. A Bayesian Hierarchical Model for Estimating National PFAS Drinking Water Occurrence. *AWWA Water Science,* 4(3):1284. *https://doi.org/10.1002/aws2.1284.*

Cakmak, S., Lukina, A., Karthikeyan, S., Atlas, E., and Dales, R. 2022. The association between blood PFAS concentrations and clinical biochemical measures of organ function and metabolism in participants of the Canadian Health Measures Survey (CHMS). *Science of the Total Environment,* 827:153900.

California Environmental Protection Agency (CalEPA). 2021. Proposed Public Health Goals for Perfluorooctanoic Acid and Perfluorooctane Sulfonic Acid in Drinking Water. Available on the internet at: *https://oehha.ca.gov/media/downloads/crnr/pfoapfosphgdraft061021.pdf.*

Canova, C., Barbieri, G., Jeddi, M.Z., Gion, M., Fabricio, A., Dapra, F., Russo, F., Fletcher, T., and Pitter, G. 2020. Associations between perfluoroalkyl substances and lipid profile in a highly exposed young adult population in the Veneto Region. *Environment International,* 145:106117.

Cantor, K.P., Lynch, C.F., Hildesheim, M.E., Dosemeci, M., Lubin, J., Alavanja, M., and Craun, G. 1998. Drinking Water Source and Chlorination Byproducts. I. Risk of Bladder Cancer. *Epidemiology,* 9(1):21–28. *http://dx.doi.org/10.1097/00001648-199801000-00007.*

Cantor, K.P., Villanueva, C.M., Silverman, D.T., Figueroa, J.D., Real, F.X., Garcia-Closas, M., Malats, N., Chanock, S., Yeager, M., Tardon, A., Garcia-Closas, R., Serra, C., Carrato, A., Castaño-Vinyals, G., Samanic, C., Rothman, N., and Kogevinas, M. 2010. Polymorphisms in GSTT1, GSTZ1, and CYP2E1, Disinfection By-products, and Risk of Bladder Cancer in Spain. *Environmental Health Perspectives,* 118(11):1545–1550. *https://doi.org/10.1289/ehp.1002206.*

Cao L., Guo Y., Chen Y., Hong J., Wu J., and Hangbiao J. 2022. Per-/polyfluoroalkyl substance concentrations in human serum and their associations with liver cancer. *Chemosphere,* 296:134083. doi: 10.1016/j.chemosphere.2022.134083. Epub 2022 February 22. PMID: 35216980.

Catelan, D., Biggeri, A., Russo, F., Gregori, D., Pitter, G., Da Re, F., Fletcher, T., and Canova, C. 2021. Exposure to Perfluoroalkyl Substances and Mortality for COVID–19: A Spatial Ecological Analysis in the Veneto Region (Italy). *International Journal of Environmental Research and Public Health,* 18(5):2734. *https://doi.org/10.3390/ijerph18052734.*

Centers for Disease Control and Prevention (CDC). 2017. User Guide to the 2017 Period/2016 Cohort Linked Birth/Infant Death Public Use File. Available on the internet at: *https://ftp.cdc.gov/pub/Health_Statistics/NCHS/Dataset_Documentation/DVS/period-cohort-linked/17PE16CO_linkedUG.pdf.*

CDC. 2018. User Guide to the 2018 Period/2017 Cohort Linked Birth/Infant Death Public Use File. Available on the internet at: *https://ftp.cdc.gov/pub/Health_Statistics/NCHS/Dataset_Documentation/DVS/period-cohort-linked/18PE17CO_linkedUG.pdf.*

Chaikind, S., and Corman, H. 1991. The Impact of Low Birthweight on Special Education Costs. *Journal of Health Economics,* 10(3):291–311. *https://doi.org/10.1016/0167-6296(91)90031-H.*

Chang, C.J., Barr, D.B., Ryan, P.B., Panuwet, P., Smarr, M.M., Liu, K., . . . and Liang, D. 2022. Per-and polyfluoroalkyl substance (PFAS) exposure, maternal metabolomic perturbation, and fetal growth in African American women: a meet-in-the-middle approach. *Environment International,* 158:106964.

Chappell, G.A., Thompson, C.M., Wolf, J.C., Cullen, J.M., Klaunig, J.E., and Haws, L.C. 2020. Assessment of the Mode of Action Underlying the Effects of GenX in Mouse Liver and Implications for Assessing Human Health Risks. *Toxicol Pathol.* 48(3):494–508. doi: 10.1177/0192623320905803. Epub 2020 Mar 6. PMID: 32138627; PMCID: PMC7153225.

Chatterji, P., Kim, D., and Lahiri, K. 2014. Birth Weight and Academic Achievement in Childhood. *Health Economics,* 23(9):1013–1035. *https://doi.org/10.1002/hec.3074.*

Chen, A., Jandarov, R., Zhou, L., Calafat, A.M., Zhang, G., Urbina, E.M., Sarac, J., Augustin, D.H., Caric, T., Bockor, L., and Petranovic, M.Z. 2019. Association of perfluoroalkyl substances exposure with cardiometabolic traits in an island population of the eastern Adriatic coast of Croatia. *Science of the Total Environment,* 683:29–36.

Cheng, W., Dastgheib, S.A., and Karanfil, T., 2005. Adsorption of dissolved natural organic matter by modified activated carbons. *Water Research, 39*(11):2281–2290.

Chili, C.A., Westerhoff, P., and Ghosh, A. 2012. GAC removal of organic nitrogen and other DBP precursors. *Journal-American Water Works Association,* 104(7):E406–E415.

Christensen, K.Y., Raymond, M., and Meiman, J. 2018. Perfluoralkyl substances and metabolic syndrome. *International Journal of Hygiene and Environmental Health,* 222(1):147–153. doi:10.1016/j.ijheh.2018.08.014.

Christensen, K., Raymond, M., Thompson, B., and Anderson, H. 2016. Perfluoroalkyl substances in older male anglers in Wisconsin. *Environment International,* 91:312–318.

Chowdhury, Z.Z., Zain, S.M., Rashid, A.K., Rafique, R.F., and Khalid, K. 2013. Breakthrough Curve Analysis for Column Dynamics Sorption of Mn(II) Ions from Wastewater by Using *Mangostana garcinia* Peel-Based Granular-Activated Carbon. *Journal of Chemistry,* 2013:959761. *https://doi.org/10.1155/2013/959761.*

Colaizy, T.T., Bartick, M.C., Jegier, B.J., Green, B.D., Reinhold, A.G., Schaefer, A.J., Bogen, D.L., Schwarz, E.B., Stuebe, A.M., Jobe, A.H., and Oh, W. 2016. Impact of Optimized Breastfeeding on the Costs of Necrotizing Enterocolitis in Extremely Low Birthweight Infants. *The Journal of Pediatrics,* 175:100–105. *https://doi.org/10.1016/j.jpeds.2016.03.040.*

Conley, J.M., Lambright, C.S., Evans, N., Medlock-Kakaley, E., Hill, D., McCord, J., Strynar, M.J., Wehmas, L.C., Hester, S., MacMillan, D.K., and Gray Jr., L.E. 2022a. Developmental toxicity of Nafion byproduct 2 (NBP2) in the Sprague-Dawley rat with comparisons to hexafluoropropylene oxide-dimer acid (HFPO–DA or GenX) and perfluorooctane sulfonate (PFOS). *Environment International,* 160:107056. doi:10.1016/j.envint.2021.107056.

Conley, J.M., Lambright, C.S., Evans, N., Medlock-Kakaley, E., Dixon, A., Hill, D., McCord, J., Strynar, M.J., Ford, J., and Gray Jr., L.E. 2022b. Cumulative maternal and neonatal effects of combined exposure to a mixture of perfluorooctanoic acid (PFOA) and perfluorooctane sulfonic acid (PFOS) during pregnancy in the Sprague-Dawley rat. *Environment International,* 170:107631. doi.org/10.1016/j.envint.2022.107631.

Conley, J.M., Lambright, C.S., Evans, N., Farraj, A.K., Smoot, J., Grindstaff, R.D., Hill, D., McCord, J., Medlock-Kakaley, E., Dixon, A., Hines, E., and Gray Jr., L.E. 2023. Dose additive maternal and offspring effects of oral maternal exposure to a mixture of three PFAS (HFPO–DA, NBP2, PFOS) during pregnancy in the Sprague-Dawley rat. *Science of the Total Environment,* 892:164609. Doi.org/10.1016/j.scitotenv.2023.164609.

Construction Industry Institute. 2001. CII Benchmarking and Metrics Analysis Results. Retrieved from *http://web.archive.org/web/20010815104859/http://www.ciibenchmarking.org/findings.htm.*

Corton, J.C., Peters, J.M., and Klaunig, J.E. 2018. The PPARα-dependent rodent liver tumor response is not relevant to humans: addressing misconceptions. *Arch Toxicol.;* 92(1):83–119. doi: 10.1007/s00204–017–2094–7. Epub 2017 December 2. PMID: 29197930; PMCID: PMC6092738.

Costello, E., Rock, S., Stratakis, N., Eckel, S.P., Walker, D.I., Valvi, D., . . . Kohli, R. 2022. Exposure to per-and polyfluoroalkyl substances and markers of liver injury: a systematic review and meta-analysis. *Environmental Health Perspectives,* 130(4):046001.

Costet, N., Villanueva, C.M., Jaakkola, J.J.K., Kogevinas, M., Cantor, K.P., King, W.D., Lynch, C.F., Nieuwenhuijsen, M.J., and Cordier, S. 2011. Water Disinfection By-products and Bladder Cancer: is there a European Specificity? A Pooled and Meta-analysis of European Case-control Studies. *Occupational & Environmental Medicine,* 68(5):379–385. *https://doi.org/10.1136/oem.2010.062703.*

Cuthbertson, A.A., Kimura, S.Y., Liberatore, H.K., Summers, R.S., Knappe, D.R., Stanford, B.D., Maness, J.C., Mulhern, R.E., Selbes, M., and Richardson, S.D. 2019. Does granular activated carbon with chlorination produce safer drinking water? From disinfection byproducts and total organic halogen to calculated toxicity. *Environmental Science & Technology, 53*(10):5987–5999.

Crittenden, J.C., Vaitheeswaran, K., Hand, D.W., Howe, E.W., Aieta, E.M., Tate, C.H., McGuire, M.J., and Davis, M.K. 1993. Removal of Dissolved Organic Carbon using Granular Activated Carbon. *Water Research,* 27(4):715–721. *https://doi.org/10.1016/0043-1354(93)90181-G.*

D'Agostino, R.B., Vasan, R.S., Pencina, M.J., Wolf, P.A., Cobain, M., Massaro, J.M., and Kannel, W.B. 2008. General Cardiovascular Risk Profile for Use in Primary Care. *Circulation,* 117(6):743–753. *https://doi.org/10.1161/CIRCULATIONAHA.107.699579.*

Dastgheib, S.A., Karanfil, T., and Cheng, W. 2004. Tailoring activated carbons for enhanced removal of natural organic matter from natural waters. *Carbon,* 42(3):547–557.

Deeks, J.J. 2002. Issues in the Selection of a Summary statistic for Meta-analysis of Clinical Trials with Binary Outcomes. *Statistics in Medicine,* 21(11):1575–1600. *https://doi.org/10.1002/sim.1188.*

DeLuca, N.M., Minucci, J.M., Mullikin, A., Slover, R., and Cohen Hubal, E.A. 2022. Human exposure pathways to poly- and perfluoroalkyl substances (PFAS) from indoor media: A systematic review. *Environ Int.,* 162:107149. doi: 10.1016/j.envint.2022.107149.

Dickenson, E., and Higgins, C. 2016. Treatment Mitigation Strategies for Poly- and Perfluorinated Chemicals. *Water Research Foundation.* Available on the internet at: *https://www.waterrf.org/research/projects/treatment-mitigation-strategies-poly-and-perfluorinated-chemicals.*

Dobson, K.G., Ferro, M.A., Boyle, M.H., Schmidt, L.A., Saigal, S., and Van Lieshout, R.J. 2018. How do Childhood Intelligence and Early Psychosocial Adversity Influence Income Attainment Among Adult Extremely Low Birth Weight Survivors? A Test of the Cognitive Reserve Hypothesis. *Development and Psychopathology,* 30(4):1421–1434. *https://doi.org/10.1017/S0954579417001651.*

Domingo, J.L., and Nadal, M. 2019. Human Exposure to Per- and Polyfluoroalkyl Substances (PFAS) through Drinking Water: A Review of the Recent Scientific Literature. *Environmental Research,* 177(2019):108648. *https://doi.org/10.1016/j.envres.2019.108648.*

Dong, Z., Wang, H., Yu, Y.Y., Li, Y.B., Naidu, R., and Liu, Y. 2019. Using 2003–2014 US NHANES data to determine the associations between per-and polyfluoroalkyl substances and cholesterol: Trend and implications. *Ecotoxicology and Environmental Safety,* 173:461–468.

Du, Z., Deng, S., Bei, Y., Huang, Q., Wang, B., Huang, J., and Yu, G. 2014. Adsorption behavior and mechanism of perfluorinated compounds on various adsorbents—a review. *Journal of Hazardous Materials,* 274:443–454. *https://doi.org/10.1016/j.jhazmat.2014.04.038*

Dunder, L., Lind, P.M., Salihovic, S., Stubleski, J., Kärrman, A., and Lind, L. 2022. Changes in plasma levels of per- and polyfluoroalkyl substances (PFAS) are associated with changes in plasma lipids-A longitudinal study over 10 years. *Environmental Research,* 211:112903.

Dzierlenga, M., Crawford, L., and Longnecker, M. 2020. Birth Weight and Perfluorooctane Sulfonic Acid: a Random-effects Meta-regression Analysis. *Environmental Epidemiology,* 4(3):e095. *https://doi.org/10.1097/EE9.0000000000000095.*

Elder, T., Figlio, D., Imberman, S., and Persico, C. 2020. The Role of Neonatal Health in the Incidence of Childhood Disability. *American Journal of Health Economics,* 6(2):216–250. *https://doi.org/10.1086/707833.*

Elmore, S.A., Dixon, D., Hailey, J.R., Harada, T., Herbert, R.A., Maronpo, R.R., Nolte, T., Rehg, J.E., Rittinghausen, S., Rosol, T.J., Satoh, H., Vidal, J.D., Willard-Mack, C.L., and Creasy, D.M. 2016. Recommendations from the INHAND apoptosis/necrosis working group. *Toxicology Pathology,* 44(2):173–88. doi:10.1177/0192623315625859.

Engels, E.A., Schmid, C.H., Terrin, N., Olkin, I., and Lau, J. 2000. Heterogeneity and Statistical Significance in Meta-analysis: an Empirical Study of 125 Meta-analyses. *Statistics in Medicine,* 19(13):1707–1728. *https://doi.org/10.1002/1097-0258(20000715)19:13<1707::aid-sim491>3.0.co;2-p.*

Eschauzier, C., Beerendonk, E., Scholte-Veenendaal, P., and De Voogt, P. 2012. Impact of treatment processes on the removal of perfluoroalkyl acids from the drinking water production chain. *Environmental Science & Technology,* 46(3):1708–1715. *https://doi.org/10.1021/es201662b.*

European Food Safety Authority (EFSA), Knutsen, H.K., Alexander, J., Barregard, L., Bignami, M., Bruschweiler, B., Ceccatelli, S., Cottrill, B., Dinovi, M., Edler, L., GraslKraupp, B., Hogstrand, C., Hoogenboom, L.R., Nebbia, C.S., Oswald, I.P., Petersen, A., Rose, M., Roudot, A.C., Vleminckx, C., Vollmer, G., Wallace, H., Bodin, L., Cravedi, J.P., Halldorsson, T.I., Haug, L.S., Johansson, N., van Loveren, H., Gergelova, P., Mackay, K., Levorato, S., van Manen, M., and Schwerdtle, T. 2018. Risk to Human Health Related to the Presence of Perfluorooctane Sulfonic Acid and Perfluorooctanoic Acid in Food. *EFSA Journal,* 16(12):e05194. *https://doi.org/10.2903/j.efsa.2018.5194.*

EFSA, Schrenk, D., Bignami, M., BodinL., Chipman, J.K., Del Mazo, J., Grasl-Kraupp, B., Hogstrang, C., Hoogenboom, L.R., Leblanc, J.C., Nebbia, C.S., Nielsen, E., Ntzani, E., Petersen, A., Sand, S., Vleminckx, C., Wallace, H., Barregard, L., Ceccatelli, S., Cravedi, J.P., Halldorsson, T.I., Haug, L.S., Johansson, N., Knutsen, H.K., Rose, M., Roudot, A.C., Van Loveren, H., Vollmer, G., Mackay, K., Riolo, F., and Schwerdtle, T. 2020. Risk to Human Health Related to the Presence of Perfluoroalkyl Substances in Food. *EFSA Journal,* 18(9):e06223. *https://doi.org/10.2903/j.efsa.2020.6223.*

Fisher, M., Arbuckle, T.E., Wade, M., and Haines, D. A. 2013. Do perfluoroalkyl substances affect metabolic function and plasma lipids?—Analysis of the 2007–2009, Canadian Health Measures Survey (CHMS) Cycle 1. *Environmental Research,* 121:95–103.

Felter, S.P., Foreman, J.E., Boobis, A., Corton, J.C., Doi, A.M., Flowers, L., Goodman, J., Haber, L.T., Jacobs, A., Klaunig, J.E., Lynch, A.M., Moggs, J., and Pandiri, A. 2018. Human relevance of rodent liver tumors: Key insights from a Toxicology Forum workshop on nongenotoxic modes of action. *Regulatory Toxicology and Pharmacology,* 92:1–7. *https://doi.org/10.1016/j.yrtph.2017.11.003.*

Feng, X., Cao, X., Zhao, S., Wang, X., Hua, X., Chen, L., and Chen, L. 2017. Exposure of pregnant mice to perfluorobutanesulfonate causes hypothyroxinemia and developmental abnormalities in female offspring. *Toxicological Sciences,* 155:409–419. *http://dx.doi.org/10.1093/toxsci/kfw219.*

Forrester, E. 2019. Calgon Carbon PFAS Experience. Calgon Carbon Corporation. Letter to EPA. March 25, 2019.

Frawley, R.P., Smith, M., Cesta, M.F., Hayes-Bouknight, S., Blystone, C., Kissling, G.E., Harris, S., and Germolec, D. 2018. Immunotoxic and hepatotoxic effects of perfluoro-n-decanoic acid (PFDA) on female Harlan Sprague–Dawley rats and B6C3F1/N mice when administered by oral gavage for 28 days. *Journal of Immunotoxicology,* 15(1):41–52.

Freeman, L.E., Cantor, K.P., Baris, D., Nuckols, J.R., Johnson, A., Colt, J.S., Schwenn, M., Ward, M.H., Lubin, J.H., Waddell, R., Hosain, G.M., Paulu, C., McCoy, R., Moore, L.E., Huang, A.T., Rothman, N., Karagas, M.R., and Silverman, D. T. 2017. Bladder Cancer and Water Disinfection By-product Exposures through Multiple Routes: A Population-Based Case-Control Study (New England, USA). *Environmental Health Perspectives,* 125(6).

Fromme, H, Tittlemier, S.A., Volkel, W., Wilhelm, M., and Twardella, D. 2009. Perfluorinated Compounds—Exposure Assessment for the General Population in Western Countries. *International Journal of Hygiene and Environmental Health,* 212:239–270. *https://doi.org/10.1016/j.ijheh.2008.04.007.*

Ghisari, M., Long, M., Røge, D.M., Olsen, J., and Bonefeld-Jørgensen, E.C. 2017. Polymorphism in xenobiotic and estrogen metabolizing genes, exposure to perfluorinated compounds and subsequent breast cancer risk: A nested case-control study in the Danish National Birth Cohort. *Environmental Research,* 154:325–333.

Gilbert, S.G., and Weiss, B. 2006. A rationale for lowering the blood lead action level from 10 to 2 microg/dL. *Neurotoxicology,* 27(5):693–701. doi: 10.1016/j.neuro.2006.06.008. Epub 2006 Aug 4. PMID: 16889836; PMCID: PMC2212280.

Gleick, P.H., and Cooley, H.S. 2009. Energy implications of bottled water. *Environ. Res. Lett.* 4(1):014009. 10.1088/1748–9326/4/1/014009.

Gleason, J.A., Post, G.B., and Fagliano, J.A. 2015. Associations of perfluorinated chemical serum concentrations and biomarkers of liver function and uric acid in the US population (NHANES), 2007–2010. *Environmental Research,* 136:8–14.

Goff, DC, Lloyd-Jones, D.M., Bennett, G., Coady, S., D'Agostino, R., Gibbons, R., Greenland, P., Lackland, D.T., Levy, D., O'donnell, C.J., Robinson, J.G., Schwartz, J.S., Shero, S.T., Smith, S.C., Sorlie, P., Stone, N., and Wilson, P.W.F. 2014. 2013 ACC/AHA Guideline on the Assessment of Cardiovascular Risk: A Report of the American College of Cardiology/American Heart Association Task Force on Practice Guidelines. *Circulation,* 129(25 supplemental 2):49–73. *https://doi.org/10.1161/01.cir.0000437741.48606.98.*

Goodrich, J.A., Walker, D., Lin, X., Wang, H., Lim, T., McConnell, R., Conti, D.V., Chatzi, L., and Setiawan, V.W. 2022. Exposure to Perfluoroalkyl Substances and Risk of Hepatocellular Carcinoma in a Multiethnic Cohort. *JHEP Reports,* 100550. *https://doi.org/10.1016/j.jhepr.2022.100550.*

Govarts, E., Iszatt, N., Trnovec, T., De Cock, M., Eggesbø, M., Murinova, L.P., van de Bor, M., Guxens, M., Chevrier, C., Koppen, G., and Lamoree, M. 2018. Prenatal exposure to endocrine disrupting chemicals and risk of being born small for gestational age: Pooled analysis of seven European birth cohorts. *Environment International,* 115:267–278. doi:10.1016/j.envint.2018.03.01.

Grandjean, P., Andersen, E.W., Budtz-Jørgensen, E., Nielsen, F., Molbank, K., Weihe, P., and Heilmann, C. 2012. Serum Vaccine Antibody Concentrations in Children Exposed to Perfluorinated Compounds. *Jama,* 307(4):391–397.

Grandjean, P., Heilmann, C., Nielsen, F., Mogensen, U.B., Timmermann, A.G., and Budtz-Jørgensen, E. 2017a. Estimated exposures to perfluorinated compounds in infancy predict attenuated vaccine antibody concentrations at age 5-years. *Journal of Immunotoxicology,* 14(1):188–195. doi:10.1080/1547691X.2017.1360968.

Grandjean, P., Heilmann, C., Weihe, P., Nielsen, F., Mogensen, U.B., and Budtz-Jørgensen, E. 2017b. Serum Vaccine Antibody Concentrations in Adolescents Exposed to Perfluorinated Compounds. *Environmental Health Perspectives,* 125(7):077018. doi:10.1289/EHP275.

Grandjean, P., Timmermann, C.A.G., Kruse, M., Nielsen, F., Vinholt, P.J., Boding, L., Heilmann, C., and Mølbak, K. 2020. Severity of COVID–19 at Elevated Exposure to Perfluorinated Alkylates. *PLoS One,* 15(12):e0244815. *https://doi.org/10.1371/journal.pone.0244815.*

Gray, L.E., Conley, J.M., and Bursian, S.J. 2023. Dose Addition Models Accurately Predict the Subacute Effects of a Mixture of Perfluorooctane Sulfonate and Perfluorooctanoic Acid on Japanese Quail (Coturnix japonica) Chick Mortality. *Environmental Toxicology and Chemistry. https://doi.org/10.1002/etc.5758.*

Greco, S.L., Belova, A., Haskell, J., and Backer, L. 2019. Estimated Burden of Disease from Arsenic in Drinking Water Supplied by Domestic Wells in the United States. *Journal of Water and Health,* 17(5):801–812. *https://doi.org/10.2166/wh.2019.216.*

Grupper, M.A.; Schreiber, M.E.; Sorice, M.G. 2021. How Perceptions of Trust, Risk, Tap Water Quality, and Salience Characterize Drinking Water Choices. *Hydrology,* 8:49. *https://doi.org/10.3390/hydrology8010049.*

Guelfo, J.L., and Adamson, D.T. 2018. Evaluation of a national data set for insights into sources, composition, and concentrations of per- and polyfluoroalkyl substances (PFASs) in U.S. drinking water. *Environmental Pollution,* 236:505–513. *https://doi.org/10.1016/j.envpol.2018.01.066.*

Hall, A.P., Elcombe, C.R., Foster, J.R., Harada, T., Kaufmann, W., Knippel, A., Küttler, K., Malarkey, D.E., Maronpot, R.R., Nishikawa, A., Nolte, T., Schulte, A., Strauss, V., and York, M.J. 2012. Liver hypertrophy: a review of adaptive (adverse and non-adverse) changes—conclusions from the 3rd international ESTP expert workshop. *Toxicologic Pathology,* 40(7):971–994. doi:10.1177/0192623312448935.

Hardell, E., Karrman, A., van Bavel, B., Bao, J., Carlberg, M., and Hardell, L. 2014. Case-control study on perfluorinated alkyl acids (PFAAs) and the risk of prostate cancer. *Environment International,* 63:35–39. doi:10.1016/j.envint.2013.10.005.

Heintz, M.M., Chappell, G.A., Thompson, C.M., and Haws, L.C. 2022. Evaluation of Transcriptomic Responses in Livers of Mice Exposed to the Short-Chain PFAS Compound HFPO–DA. *Frontiers in Toxicology,* 4:937168. *https://doi.org/10.3389/ftox.2022.937168.*

Heintz, M.M., Haws, L.C., Klaunig, J.E., Cullen, J.M., and Thompson, C.M. 2023. Assessment of the mode of action underlying development of liver lesions in mice following oral exposure to HFPO–DA and relevance to humans. *Toxicological Sciences,* 192(1):15–29. *https://doi.org/10.1093/toxsci/kfad004.*

Hines, C.T., Padilla, C.M., and Ryan, R.M. 2020. The Effect of Birth Weight on Child Development Prior to School Entry. *Child Development,* 91(3):724–732. *https://doi.org/10.1111/cdev.13355.*

Hoff, P.D. 2009. A first course in Bayesian statistical methods (Vol. 580). New York: Springer.

Huang, M., Jiao, J., Zhuang, P., Chen, X., Wang, J., and Zhang, Y. 2018. Serum polyfluoroalkyl chemicals are associated with risk of cardiovascular diseases in national US population. *Environment International,* 119:37–46.

International Agency for Research on Cancer (IARC). 2016. Some chemicals used as solvents and in polymer manufacture. Perfluorooctanoic acid. In IARC Monographs on the Evaluation of the Carcinogenic Risk of Chemicals to Humans. Lyons, France: World Health Organization. *https://publications.iarc.fr/547*.

ICF. 2020. Memorandum re: Trends in Infant Mortality by Birth Weight Categories, EPA Contract No. 68HE0C18D0001, TO 68HERC20F0107. Submitted to U.S. Environmental Protection Agency Office of Groundwater and Drinking Water, July 30, 2020.

ICF. 2021. Memorandum re: Summary of Literature Review of Non-Medical Effects and Economic Burden Related to Low Birth Weight Infants. Submitted to U.S. Environmental Protection Agency Office of Groundwater and Drinking Water, October 5, 2021.

Ioannou, G.N., Boyko, E.J., and Lee, S.P. 2006a. The prevalence and predictors of elevated serum aminotransferase activity in the United States in 1999–2002. *The American College of Gastroenterology,* 101(1):76–82.

Ioannou, G.N., Weiss, N.S., Boyko, E.J., Mozaffarian, D., and Lee, S.P. 2006b. Elevated serum alanine aminotransferase activity and calculated risk of coronary heart disease in the United States. *Hepatology,* 43(5):1145–1151.

Iriarte-Velasco, U., Álvarez-Uriarte, J.I., Chimeno-Alanis, N., and González-Velasco, J.R., 2008. Natural organic matter adsorption onto granular activated carbons: implications in the molecular weight and disinfection byproducts formation. *Industrial & Engineering Chemistry Research,* 47(20):7868–7876.

Jain, R.B., and Ducatman, A. 2019. Selective associations of recent low concentrations of perfluoroalkyl substances with liver function biomarkers: NHANES 2011 to 2014 data on US adults aged ≥20 years. *Journal of Occupational and Environmental Medicine,* 61(4):293–302.

Jelenkovic, A., Mikkonen, J., Martikainen, P., Latvala, A., Yokoyama, Y., Sund, R., Vuoksimaa, E., Rebato, E., Sung, J., Kim, J., Lee, J., Lee, S., Stazi, M.A., Fagnani, C., Brescianini, S., Derom, C.A., Vlietinck, R.F., Loos, R.J.F., Krueger, R.F., McGue, M., Pahlen, S., Nelson, T.L., Whitfield, K.E., Brandt, I., Nilsen, T.S., Harris, J.R., Cutler, T.L., Hopper, J.L., Tarnoki, A.D., Tarnoki, D.L., Sørensen, T.I.A., Kaprio, J., and Silventoinen, K. 2018. Association Between Birth Weight and Educational Attainment: an Individual-based Pooled Analysis of Nine Twin Cohorts. *Journal of Epidemiology & Community Health,* 72(9):832–837. *https://doi.org/10.1136/jech-2017-210403*.

Ji, J., Song, L., Wang, J., Yang, Z., Yan, H., Li, T., Yu, L., Jian, L., Jiang, F., Li, J., Zheng, J., and Li, K. 2021. Association Between Urinary Per- and Poly-Fluoroalkyl Substances and COVID–19 Susceptibility. *Environment International,* 153(2021):106524. *https://doi.org/10.1016/j.envint.2021.106524*.

Jiang, J., Zhang, X., Zhu, X., and Li, Y. 2017. Removal of intermediate aromatic halogenated DBPs by activated carbon adsorption: a new approach to controlling halogenated DBPs in chlorinated drinking water. *Environmental Science & Technology,* 51(6):3435–3444.

Joyce, C., Goodman-Bryan, M., and Hardin, A. 2012. Preterm Birth and Low Birth Weight. Available on the internet at: *http://www.urbanchildinstitute.org/sites/all/files/2010-10-01-PTB-and-LBW.pdf*.

Kamai, E.M., McElrath, T.F., and Ferguson, K.K. 2019. Fetal Growth in Environmental Epidemiology: Mechanisms, Limitations, and a Review of Associations with Biomarkers of Non-persistent Chemical Exposures during Pregnancy. *Environmental Health,* 18(1):43. *https://doi.org/10.1186/s12940-019-0480-8*.

Khalil, N., Chen, A., Lee, M., Czerwinski, S.A., Ebert, J.R., DeWitt, J.C., and Kannan, K. 2016. Association of perfluoroalkyl substances, bone mineral density, and osteoporosis in the US population in NHANES 2009–2010. *Environmental Health Perspectives,* 124(1):81–87.

Khalil, N., Ebert, J.R., Honda, M., Lee, M., Nahhas, R.W., Koskela, A., Hangartner, T., and Kannan, K. 2018. Perfluoroalkyl substances, bone density, and cardio-metabolic risk factors in obese 8–12 year old children: A pilot study. *Environmental Research,* 160:314–321. doi:10.1016/j.envres.2017.10.014.

Khera, R., Ransom, P., and Speth, T.F. 2013. Using work breakdown structure models to develop unit treatment costs. *Journal-American Water Works Association, 105*(11):E628–E641.

Kim, W.H., Nishijima, W., Shoto, E., and Okada, M. 1997. Competitive Removal of Dissolved Organic Carbon by Adsorption and Biodegradation on Biological Activated Carbon. *Water Science and Technology,* 35(7):147–153. *https://doi.org/10.1016/S0273-1223(97)00125#-X*.

Kim, W.R., Flamm, S.L., Di Bisceglie, A.M., and Bodenheimer, H.C. 2008. Serum activity of alanine aminotransferase (ALT) as an indicator of health and disease. *Hepatology,* 47(4):1363–1370.

King, W.D., and Marrett, L.D. 1996. Case-control study of bladder cancer and chlorination by-products in treated water (Ontario, Canada). *Cancer Causes & Control,* 7(6):596–604. doi:*https://doi.org/10.1007/BF00051702*.

Kirisits, M.J., Snoeyink, V.L., and Kruithof, J.C. 2000. The reduction of bromate by granular activated carbon. *Water Research,* 34(17):4250–4260.

Klein, R., and Lynch, M. 2018. Development of Medical Cost Estimates for Adverse Birth Outcomes. Prepared for U.S. EPA National Center for Environmental Economics.

Kowlessar, N.M., Jiang, H.J., and Steiner, C. 2013. Hospital Stays for Newborns, 2011: Statistical Brief #163. Available on the internet at: *https://pubmed.ncbi.nlm.nih.gov/24308074/*.

Kwo, P.Y., Cohen, S.M., and Lim, J.K. 2017. ACG clinical guideline: evaluation of abnormal liver chemistries. *The American College of Gastroenterology,* 112(1):18–35.

Lauritzen, H.B., Larose, T.L., Øien, T., Sandanger, T.M., Odland, J.O., van der Bor, M., and Jacobsen, G.W. 2017. Maternal serum levels of perfluoroalkyl substances and organochlorines and indices of fetal growth: a Scandinavian case—cohort study. *Pediatric Research,* 81(1):33–42. doi:10.1038/pr.2016.187.

Lenters, V., Portengen, L., Rignell-Hydbom, A., Jonsson, B.A.G., Lindh, C.H., Piersma, A.H., Toft, G., Bonde, J.P., Heederik, D., Rylander, L., and Vermeulen, R. 2016. Prenatal Phthalate, Perfluoroalkyl Acid, and Organochlorine Exposures and Term Birth Weight in Three Birth Cohorts: Multi-Pollutant Models Based on Elastic Net Regression. *Environmental Health Perspectives,* 124:365–372. *https://doi.org/10.1289/ehp.1408933*.

Levêque, J.G., and Burns, R.C. 2017. A Structural Equation Modeling approach to water quality perceptions. *J Environ Manage.,* 197:440–447. 10.1016/j.jenvman.2017.04.024. Epub 2017 April 12. PMID: 28411571.

Li, N., Liu, Y., Papandonatos, G.D., Calafat, A.M., Eaton, C.B., Kelsey, K.T., Cecil, K.M., Kalkwarf, H.J., Yolton, K., Lanphear, B.P., Chen, A., and Braun, J.M. 2021. Gestational and childhood exposure to per- and polyfluoroalkyl substances and cardiometabolic risk at age 12 years. *Environment International,* 147:106344.

Li, S., Peng, Y., Wang, X., Qian, Y., Xiang, P., Wade, SW, Guo, H., Lopez, J.A.G., Herzog, C.A., and Handelsman, Y. 2019. Cardiovascular Events and Death after Myocardial Infarction or Ischemic Stroke in an Older Medicare Population. *Clinical Cardiology,* 42(3):391–399. *https://doi.org/10.1002/clc.23160*.

Liao, S., Yao, W., Cheang, I., Tang, X., Yin, T., Lu, X., Zhou, Y., Zhang, H., and Li, X. 2020. Association between Perfluoroalkyl Acids and the Prevalence of Hypertension among US Adults. *Ecotoxicology and Environmental Safety,* 196(2020):110589. *https://doi.org/10.1016/j.ecoenv.2020.110589*.

Liew, Z., Olsen, J., Cui, X., Ritz, B., and Arah, O.A. 2015. Bias from conditioning on live birth in pregnancy cohorts: an illustration based on neurodevelopment in children after prenatal exposure to organic pollutants. *International Journal of Epidemiology,* 44(1):345–354.

Lin, P.D., Cardenas, A., Hauser, R., Gold, D.R., Kleinman, K. P., Hivert, M.-F., Fleisch, A.F., Calafat, A.M., Webster, T.F., Horton, E.S., and Oken, E. 2019. Per- and polyfluoroalkyl substances and blood lipid levels in pre-diabetic adults—longitudinal analysis of the diabetes prevention program outcomes study. *Environment International,* 129:343–353.

Lin, P.D., Cardenas, A., Hauser, R., Gold, D.R., Kleinman, K.P., Hivert, M.-F., Calafat, A.M., Webster, T.F., Horton, E.S.

and Oken, E. 2020. Per- and polyfluoroalkyl substances and blood pressure in pre-diabetic adults—cross-sectional and longitudinal analyses of the diabetes prevention program outcomes study. *Environment International,* 137:105573.

Liu, G., Dhana, K., Furtado, J.D., Rood, J., Zong, G., Liang, L., Qi, L., Bray, G.A., DeJonge, L., Coull, B., and Grandjean, P. 2018. Perfluoroalkyl substances and changes in body weight and resting metabolic rate in response to weight-loss diets: a prospective study. *PLoS Medicine,* 15(2):e1002502.

Liu, G., Zhang, B., Hu, Y., Rood, J., Liang, L., Qi, L., Bray, G.A., DeJonge, L., Coull, B., Grandjean, P., and Furtado, J.D. 2020. Associations of perfluoroalkyl substances with blood lipids and apolipoproteins in lipoprotein subspecies: the POUNDS-lost study. *Environmental Health,* 19(1):1–10.

Liu, J.J., Cui, X.X., Tan, Y.W., Dong, P.X., Ou, Y.Q., Li, Q.Q., Chu, C., Wu, L.Y., Liang, L.X., Qin, S.J., Zeeshan, M., Zhou, Y., Hu, L.W., Liu, R.Q., Zeng, X.W., Dong, G.H., and Zhao, X.M. 2022. Per- and perfluoroalkyl substances alternatives, mixtures and liver function in adults: A community-based population study in China. *Environmental International,* 163:107179. *http://dx.doi.org/10.1016/j.envint.2022.107179.*

Liu, M., Zhang, G., Meng, L., Han, X., Li, Y., Shi, Y., Li, A., Turyk, M.E., Zhang, Q., and Jiang, G. 2021. Associations between novel and legacy per- and polyfluoroalkyl substances in human serum and thyroid cancer: a case and healthy population in Shandong Province, East China. *Environmental Science & Technology,* 56(10):6144–6151.

Liu, Z., Que, S., Xu, J., and Peng, T. 2014. Alanine aminotransferase-old biomarker and new concept: a review. *International Journal of Medical Sciences,* 11(9):925.

Li, Y., Fletcher, T., Mucs, D., Scott, K., Lindh, C.H., Tallving, P., and Jakobsson, K. 2018. Half-lives of PFOS, PFHxS and PFOA after end of exposure to contaminated drinking water. *Occupational and Environmental Medicine,* 75(1):46–51.

Lloyd-Jones, D.M., Huffman, M.D., Karmali, K.N., Sanghavi, D.M., Wright, J.S., Pelser, C., Gulati, M., Masoudi, F.A., and Goff, Jr., DC 2017. Estimating Longitudinal Risks and Benefits from Cardiovascular Preventive Therapies among Medicare Patients: the Million Hearts Longitudinal ASCVD Risk Assessment Tool: a Special Report from the American Heart Association and American College of Cardiology. *Circulation,* 135(13):e79–e813. *https://doi.org/10.1016/j.jacc.2016.10.018.*

Louis, G.M.B., Zhai, S., Smarr, M.M., Grewal, J., Zhang, C., Grantz, K.L., Hinkle, S.N., Sundaram, R., Lee, S., Honda, M., and Oh, J. 2018. Endocrine disruptors and neonatal anthropometry, NICHD fetal growth studies-singletons. *Environment International,* 119:515–526.

Lu, S., and Bartell, S.M. 2020. Serum PFAS Calculator for Adults. Available on the internet at: *www.ics.uci.edu/~sbartell/pfascalc.html.*

Luo, D., Wu, W.X., Pan, Y.A., Du, B.B., Shen, M.J., and Zeng, L.X. 2021. Associations of prenatal exposure to per- and polyfluoroalkyl substances with the neonatal birth size and hormones in the growth hormone/insulin-like growth factor axis. *Environmental Science & Technology,* 55:11859–11873. *http://dx.doi.org/10.1021/acs.est.1c02670.*

Ma, S., and Finch, B.K. 2010. Birth Outcome Measures and Infant Mortality. *Population Research and Policy Review,* 29(6):865–891. *https://doi.org/10.1007/s11113-009-9172-3.*

Mahoney, H., Xie, Y., Brinkmann, M., and Giesy, J.P. 2022. Next Generation Per- and Poly-Fluoroalkyl Substances: Status and Trends, Aquatic Toxicity, and Risk Assessment. *Eco-Environment & Health,* 1(2):117–131. *https://doi.org/10.1016/j.eehl.2022.05.002.*

Manzano-Salgado, C.B., Casas, M., Lopez-Espinosa, M.-J., Ballester, F., Iñiguez, C., Martinez, D., Romaguera, D., Fernández-Barrés, S., Santa-Marina, L., Basterretxea, M., and Schettgen, T. 2017. Prenatal exposure to perfluoroalkyl substances and cardiometabolic risk in children from the Spanish INMA Birth Cohort Study. *Environmental Health Perspectives,* 125(9):097018.

Martin, J.J., Winslow, S.D., and Munch, D.J. 2007. A New Approach to Drinking-Water-Quality Data: Lowest-Concentration Minimum Reporting Level. *Environmental Science & Technology,* 41(3):677–681. *https://doi.org/10.1021/es072456n.*

Martin, O., Scholze, M., Ermler, S., McPhie, J., Bopp, S.K., Kienzler, A., Parissis, N., and Kortenkamp, A. 2021. Ten years of research on synergisms and antagonisms in chemical mixtures: a systematic review and quantitative reappraisal of mixture studies. *Environment International,* 146:106206. *https://doi.org/10.1016/j.envint.2020.106206.*

Mastropietro, T.F., Bruno, R., Pardo, E., and Armentano, D. 2021. Reverse Osmosis and Nanofiltration Membranes for Highly Efficient PFASs Removal: Overview, Challenges and Future Perspectives. *Dalton Transactions,* 50(16):5398–5410. *https://doi.org/10.1039/d1dt00360g.*

Mathiesen, U., Franzen, L., Frydén, A., Foberg, U., and Bodemar, G. 1999. The clinical significance of slightly to moderately increased liver transaminase values in asymptomatic patients. *Scandinavian Journal of Gastroenterology,* 34(1):85–91.

Mattsson, K., Rignell-Hydbom, A., Holmberg, S., Thelin, A., Jönsson, B.A., Lindh, C.H., Sehlstedt, A., and Rylander, L. 2015. Levels of perfluoroalkyl substances and risk of coronary heart disease: Findings from a population-based longitudinal study. *Environmental Research,* 142:148–154.

McCleaf, P., Englund, S., Östlund, A., Lindegren, K., Wiberg, K., and Ahrens, L. 2017. Removal Efficiency of Multiple Poly- and Perfluoroalkyl Substances (PFASs) in Drinking Water using Granular Activated Carbon (GAC) and Anion Exchange (AE) Column Tests. *Water Research,* 120(2017):77–87. *https://doi.org/10.1016/j.watres.2017.04.057.*

McCullough, K., Lytle, D.A., and Sorg, T.J. 2005. *Treatment Technologies for Arsenic Removal.* EPA/600/S–05/006. Available on the internet at *https://nepis.epa.gov/Exe/ZyPDF.cgi/20017IDW.PDF?Dockey=20017IDW.PDF.*

Messner, M.J., Chappell, C.L., and Okhuysen, P.C. 2001. Risk assessment for Cryptosporidium: a hierarchical Bayesian analysis of human dose response data. *Water Research,* 35(16):3934–3940. *https://doi.org/10.1016/S0043-1354(01)00119-1.*

Mi, X.; Yang, Y.Q.; Zeeshan, M.; Wang, Z.B.; Zeng, X.Y.; Zhou, Y.; Yang, B.Y.; Hu, L.W.; Yu, H.Y.; Zeng, X.W.; Liu, R.Q.; Dong, G.H. 2020. Serum levels of per- and polyfluoroalkyl substances alternatives and blood pressure by sex status: Isomers of C8 health project in China. *Chemosphere,* 261:127691. *https://hero.epa.gov/hero/index.cfm/reference/details/reference_id/6833736.*

Mitro, S.D.; Sagiv, S.K.; Fleisch, A.F.; Jaacks, L.M.; Williams, P.L.; Rifas-Shiman, S.L.; Calafat, A.M.; Hivert, M.F.; Oken, E.; James-Todd, T.M. 2020. Pregnancy per- and polyfluoroalkyl substance concentrations and postpartum health in project viva: A prospective cohort. *J Clin Endocrinol Metab,* 105:e3415–e3426. *https://hero.epa.gov/hero/index.cfm/reference/details/reference_id/6833625.*

National Academies of Sciences, Engineering, and Medicine (NASEM). 2022. Guidance on PFAS Exposure, Testing, and Clinical Follow-Up. Washington, DC: The National Academies Press. *https://doi.org/10.17226/26156.*

National Drinking Water Advisory Council (NDWAC). 2022. National Drinking Water Advisory Council Fall 2022 Meeting Summary Report. **Federal Register**. 87 FR 18016 (March 29, 2022), *https://www.federalregister.gov/documents/2022/03/29/2022-06576/meeting-of-the-national-drinking-water-advisory-council.*

National Drinking Water Advisory Council (NDWAC). 2023. National Drinking Water Advisory Council Meeting Summary Report. *https://www.epa.gov/system/files/documents/2024-02/ndwac-meeting-summary-august-2023_0.pdf.*

National Institutes of Health (NIH). 2017. What are the risks of preeclampsia & eclampsia to the fetus? Retrieved from *https://www.nichd.nih.gov/health/topics/preeclampsia/conditioninfo/risk-fetus.*

NIH. 2018. What are the risks of preeclampsia & eclampsia to the mother? Retrieved from *https://www.nichd.nih.gov/health/topics/preeclampsia/conditioninfo/risk-mother.*

National Research Council (NRC). 2008. *Phthalates and Cumulative Risk Assessment: The Tasks Ahead.* The National Academies Press, Washington, DC. doi:10.17226/12528.

National Toxicology Program (NTP). 2018a. 28-day evaluation of the toxicity (C06100) of perfluorohexane sulfonate potassium salt (PFHSKslt) (3871–99–6) on Harlan Sprague-Dawley rats exposed via gavage.

NTP. 2018b. 28-day evaluation of the toxicity (C20615) of perfluorodecanoic acid (PFDA) (335–76–2) on Harlan Sprague-Dawley rats exposed via gavage [NTP]. U.S. Department of Health and Human Services. Retrieved from *http://dx.doi.org/10.22427/NTP-DATA-002-02652-0004-0000-1*.

NTP. 2021. Report on Carcinogens, Fifteenth Edition.; Research Triangle Park, NC: U.S. Department of Health and Human Services, Public Health Service.

Negri, E., Metruccio, F., Guercio, V., Tosti, L., Benfenati, E., Bonzi, R., La Vecchia, C., and Moretto, A. 2017. Exposure to PFOA and PFOS and Fetal Growth: a Critical Merging of Toxicological and Epidemiological Data. *Critical Reviews in Toxicology,* 47(6):489–515. *https://doi.org/10.1080/10408444.2016.1271972*.

Nian, M., Li, Q.-Q., Bloom, M., Qian, Z.M., Syberg, K.M., Vaughn, M.G., Wang, S.Q., Wei, Q., Zeeshan, M., Gurram, N., and Chu, C. 2019. Liver function biomarkers disorder is associated with exposure to perfluoroalkyl acids in adults: Isomers of C8 Health Project in China. *Environmental Research,* 172:81–88. *https://doi.org/10.1016/j.envres.2019.02.013*.

Nicoletti, C., Salvanes, K.G., and Tominey, E. 2018. Response of Parental Investments to Child's Health Endowment at Birth. *Health Econometrics (Contributions to Economic Analysis),* Emerald Publishing Limited, 294:175–199. *https://doi.org/10.1108/S0573-855520180000294009*.

New Jersey Drinking Water Quality Institute (NJDWQI). 2017. Maximum Contaminant Level Recommendation for Perfluorooctanoic Acid in Drinking Water. *https://www.nj.gov/dep/watersupply/pdf/pfoa-recommend.pdf*.

NJDWQI. 2018. Health-based maximum contaminant level support document: Perfluorooctane sulfonate (PFOS): Appendix A. *https://www.nj.gov/dep/watersupply/pdf/pfos-recommendation-appendix-a.pdf*.

Obsekov, V., Kahn, L.G., and Trasande, L. 2023. Leveraging systematic Reviews to Explore disease burden and costs of per- and polyfluoroalkyl substance exposures in the United States. *Exposure and Health, 15*(2):373–394.

Ochoa-Herrera, V., and Sierra-Alvarez, R. 2008. Removal of perfluorinated surfactants by sorption onto granular activated carbon, zeolite and sludge. *Chemosphere,* 72(10):1588–1593. *https://doi.org/10.1016/j.chemosphere.2008.04.029*.

Office of Management and Budget (OMB). 1977. Statistical Policy Directive 15: Standards for Maintaining, Collecting, and Presenting Federal Data on Race and Ethnicity. **Federal Register**. 62 FR 58788. May 12, 1977.

OMB. 2003. Circular A–4: Regulatory Analysis. Washington, DC: OMB. Available on the internet at: *https://obamawhitehouse.archives.gov/omb/circulars_a004_a-4/*.

OMB. 2023. Circular No. A–4. Regulatory Analysis. Washington, DC: OMB. Available on the internet at: *https://www.whitehouse.gov/wp-content/uploads/2023/11/CircularA-4.pdf*.

Osuchukwu, O., and Reed, D. 2022. Small for Gestational Age. *StatPearls.*

O'Sullivan, A.K., Rubin, J., Nyambose, J., Kuznik, A., Cohen, D.J., and Thompson, D. 2011. Cost Estimation of Cardiovascular Disease Events in the US. *Pharmacoeconomics,* 29(8);693–704. *https://doi.org/10.2165/11584620-000000000-00000*.

Papadopoulou, E., Stratakis, N., Basagana, X., Brantsæter, A.L., Casas, M., Fossati, S., Gražulevičienė, R., Haug, L.S., Heude, B., Maitre, L., and McEachan, R.R. 2021. Prenatal and postnatal exposure to PFAS and cardiometabolic factors and inflammation status in children from six European cohorts. *Environment International,* 157:106853. *https://doi.org/10.1016/j.envint.2021.106853*.

Park, J.H., Choi, J., Jun, D.W., Han, SW, Yeo, Y.H., and Nguyen, M.H. 2019. Low alanine aminotransferase cut-off for predicting liver outcomes; a nationwide population-based longitudinal cohort study. *Journal of Clinical Medicine,* 8(9):1445. *https://doi.org/10.3390/jcm8091445*.

Pitter, G., Zare Jeddi, M., Barbieri, G., Gion, M., Fabricio, A.S., Daprà, F., Russo, F., Fletcher, T., and Canova, C. 2020. Perfluoroalkyl substances are associated with elevated blood pressure and hypertension in highly exposed young adults. *Environmental Health,* 19(1):1–11. *https://doi.org/10.1186/s12940-020-00656-0*.

Ramhøj, L., Hass, U., Boberg, J., Scholze, M., Christiansen, S., Nielsen, F., and Axelstad, M. 2018. Perfluorohexane sulfonate (PFHxS) and a mixture of endocrine disrupters reduce thyroxine levels and cause antiandrogenic effects in rats. *Toxicological Sciences,* 163(2):579–591. *https://doi.org/10.1093/toxsci/kfy055*.

Ramhøj, L., Hass, U., Gilbert, M.E., Wood, C., Svingen, T., Usai, D., Vinggaard, A.M., Mandrup, K., and Axelstad, M. 2020. Evaluating thyroid hormone disruption: investigations of long-term neurodevelopmental effects in rats after perinatal exposure to perfluorohexane sulfonate (PFHxS). *Scientific Reports,* 10(1):2672. *https://doi.org/10.1038/s41598-020-59354-z*.

Regli, S., Chen, J., Messner, M., Elovitz, M.S., Letkiewicz, F.J., Pegram, R.A., Pepping, T.J., Richardson, S.D., and Wright, J.M. 2015. Estimating Potential Increased Bladder Cancer Risk Due to Increased Bromide Concentrations in Sources of Disinfected Drinking Waters. *Environmental Science & Technology,* 49(22):13094–13102. *https://doi.org/10.1021/acs.est.5b03547*.

Roth, K., Yang, Z., Agarwal, M., Liu, W., Peng, Z., Long, Z., Birbeck, J., Westrick, J., Liu, W., and Petriello, M.C. 2021. Exposure to a mixture of legacy, alternative, and replacement per-and polyfluoroalkyl substances (PFAS) results in sex-dependent modulation of cholesterol metabolism and liver injury. *Environment International,* 157:106843. *https://doi.org/10.1016/j.envint.2021.106843*.

Rubel, Jr., F. 2003. Design Manual: Removal of Arsenic from Drinking Water by Adsorptive Media. EPA 600/R–03/019. Available on the internet at *https://nepis.epa.gov/Exe/ZyPDF.cgi/30002K50.PDF?Dockey=30002K50.PDF*.

Rücker, G., Schwarzer, G., Carpenter, J., and Olkin, I. 2009. Why Add Anything to Nothing? The Arcsine Difference as a Measure of Treatment Effect in Meta-analysis with Zero Cells. *Statistics in Medicine,* 28(5):721–738. *https://doi.org/10.1002/sim.3511*.

Sagiv, S.K., Rifas-Shiman, S.L., Fleisch, A.F., Webster, T.F., Calafat, A.M., Ye, X., Gillman, M.W., and Oken, E. 2018. Early Pregnancy Perfluoroalkyl Substance Plasma Concentrations and Birth Outcomes in Project Viva: Confounded by Pregnancy Hemodynamics? *American Journal of Epidemiology,* 187:793–802. *https://doi.org/10.1093%2Faje%2Fkwx332*.

Salihovic, S., Stubleski, J., Kärrman, A., Larsson, A., Fall, T., Lind, L., and Lind, P. M. 2018. Changes in markers of liver function in relation to changes in perfluoroalkyl substances-a longitudinal study. *Environment International,* 117:196–203. *https://doi.org/10.1016/j.envint.2018.04.052*.

Shearer, J.J., Callahan, C.L., Calafat, A.M., Huang, W.Y., Jones, R.R., Sabbisetti, V.S., Freedman, N.D., Sampson, J.N., Silverman, D.T., Purdue, M.P., and Hofmann, J.N. 2021. Serum Concentrations of Per- and Polyfluoroalkyl Substances and Risk of Renal Cell Carcinoma. *Journal of the National Cancer Institute,* 113(5):580–587. *https://doi.org/10.1093/jnci/djaa143*.

Sorlini, S., and Collivignarelli, C. 2005. Chlorite removal with granular activated carbon. *Desalination,* 176(1–3):255–265. *https://doi.org/10.1016/j.desal.2004.10.030*.

Sörengård, M., Östblom, E., Köhler, S., and Ahrens, L. 2020. Adsorption Behavior of Per-and Polyfluoralkyl Substances (PFASs) to 44 Inorganic and Organic Sorbents and Use of Dyes as Proxies for PFAS Sorption. *Journal of Environmental Chemical Engineering,* 8(3):103744. *https://doi.org/10.1016/j.jece.2020.103744*.

Souza, M.C.O., Saraiva, M.C.P., Honda, M., Barbieri, M.A., Bettiol, H., Barbosa, F., and Kannan, K. 2020. Exposure to per- and polyfluorinated alkyl substances in pregnant Brazilian women and its association with fetal growth. Environmental research, 187:109585. *https://doi.org/10.1016/j.envres.2020.109585*.

Starling, A.P., Adgate, J.L., Hamman, R.F., Kechris, K., Calafat, A.M., Ye, X., and Dabelea, D. 2017. Perfluoroalkyl substances during pregnancy and offspring weight and adiposity at birth: examining mediation by maternal fasting glucose in the Healthy Start Study. *Environ Health Perspectives,* 125:067016–067011. doi:10.1289/EHP641.

Steenland K., Hofmann J.N., Silverman D.T., and Bartell S.M. 2022. Risk assessment for PFOA and kidney cancer based on a pooled analysis of two studies. *Environment International,* 167:107425. *https://pubmed.ncbi.nlm.nih.gov/35905598/.*

Steenland, K., Tinker, S., Frisbee, S., Ducatman A., and Vaccarino, V. 2009. Association of Perfluorooctanoic Acid and Perfluorooctane Sulfonate with Serum Lipids among Adults Living Near a Chemical Plant. *American Journal of Epidemiology,* 170:1269–1278. *https://doi.org/10.1093/aje/kwp279.*

Steenland, K., Zhao, L., and Winquist, A. 2015. A Cohort Incidence Study of Workers Exposed to Perfluorooctanoic Acid (PFOA). *Occupational & Environmental Medicine,* 72:373–380. *http://dx.doi.org/10.1136/oemed-2014-102364.*

Steenland, K., Barry, V., and Savitz, D. 2018. An Updated Meta-Analysis of the Association of Serum PFOA and Birthweight, with an Evaluation of Potential Biases. Paper presented at the ISEE Conference Abstracts.

Summers, R.S., Kim, S.M., Shimabuku, K., Chae, S.H. and Corwin, C.J. 2013. Granular activated carbon adsorption of MIB in the presence of dissolved organic matter. *Water Research,* 47(10):3507–3513. *https://doi.org/10.1016/j.watres.2013.03.054.*

Surveillance Research Program, National Cancer Institute. 2020a. SEER*Stat software Incidence—SEER Research Limited-Field Data, 21 Registries, November 2020 Sub (2000–2018). Available on the internet at: *seer.cancer.gov/seerstat.*

Surveillance Research Program, National Cancer Institute. 2020b. SEER*Stat software Incidence-Based Mortality—SEER Research Data, 18 Registries, November 2020 Sub (2000–2018). Available on the internet at: *seer.cancer.gov/seerstat.*

Temple, J.A., Reynolds, A.J., and Arteaga, I. 2010. Low Birth Weight, Preschool Education, and School Remediation. *Education and Urban Society,* 42(6):705–729. *https://doi.org/10.1177/0013124510370946.*

Terry, L.G., and Summers, R.S. 2018. Biodegradable organic matter and rapid-rate biofilter performance: A review. *Water Research,* 128:234–245. *https://doi.org/10.1016/j.watres.2017.09.048.*

Thom, T., Kannel, W., Silbershatz, H., and D'Agostino, R. 2001. Cardiovascular Diseases in the United States and Prevention Approaches. Hurst's the heart, 1:3–17.

Thompson, C.M., Heintz, M.M., Wolf, J.C., Cheru, R., Haws, L.C., and Cullen, J.M. 2023. Assessment of Mouse Liver Histopathology Following Exposure to HFPO–DA With Emphasis on Understanding Mechanisms of Hepatocellular Death. *Toxicologic Pathology,* 51(1–2):4–14. *https://doi.org/10.1177/01926233231159078.*

Tsai, M., Chang, S., Kuo, W., Kuo, C., Li, S., Wang, M., Chang, D.Y., Lu, Y.S., Huang, C.S., Cheng, A.L., Lin, C.H., and Chen, P. 2020. A case-control study of perfluoroalkyl substances and the risk of breast cancer in Taiwanese women. *Environment International.* doi:10.1016/j.envint.2020.105850.

United States Bureau of Labor Statistics (USBLS). 2010. BLS Handbook of Methods: The Producer Price Index. Last updated 10 July. Retrieved from *http://www.bls.gov/opub/hom/pdf/homch14.pdf.*

USBLS. 2022. Table 1. Employment Cost Index for total compensation, by occupational group and industry. Employment Cost Index Historical Listing—Volume III. Retrieved from *https://www.bls.gov/web/eci/eci-current-nominal-.dollar.pdf.*

United States Census Bureau. 2020. Annual County Resident Population Estimates by Age, Sex, Race, and Hispanic Origin: April 1, 2010 to July 1, 2019, downloaded at *https://www2.census.gov/programs-surveys/popest/* on May 3, 2022.

United States Census Bureau. 2022. *American Community Survey 5-Year Data (2009–2021).* Retrieved from *https://www.census.gov/data/developers/data-sets/acs-5year.html.*

United States Energy Information Administration (USEIA). 2023. Electricity Explained: Use of Electricity. Retrieved from *https://www.eia.gov/energyexplained/electricity/use-of-electricity.php.*

United States Environmental Protection Agency (USEPA). 1979. National Interim Primary Drinking Water Regulations; Control of Trihalomethanes in Drinking Water. **Federal Register**. 44 FR 68624. November 29, 1979.

USEPA. 1986. *Guidelines for the Health Risk Assessment of Chemical Mixtures.* EPA 630–R–98–002. Available on the internet at: *https://www.epa.gov/risk/guidelines-health-risk-assessment-chemical-mixtures.*

USEPA. 1987. National Primary Drinking Water Regulations—Synthetic Organic Chemicals; Monitoring for Unregulated Contaminants; Final Rule. **Federal Register**. 52 FR 25690, July 8, 1987.

USEPA. 1989. Notice of Availability of Drinking Water Health Advisories for pesticides. **Federal Register**. 54 FR 7599. February 22, 1989.

USEPA. 1991a. *Guidelines for Developmental Toxicity Risk Assessment.* EPA/600/FR–91/001. December 1991. *https://www.epa.gov/sites/default/files/2014-11/documents/dev_tox.pdf.*

USEPA. 1991b. *Risk Assessment Guidance for Superfund, Vol 1: Human Health Evaluation Manual (Part B, Development of Risk-Based Preliminary Remediation Goals).* EPA/540/R–92/003. EPA, Emergency and Remedial Response, Washington, DC. *https://www.epa.gov/risk/risk-assessment-guidance-superfund-rags-part-b.*

USEPA. 1991c. National Primary Drinking Water Regulations—Synthetic Organic Chemicals and Inorganic Chemicals; Monitoring for Unregulated Contaminants; National Primary Drinking Water Regulations Implementation; National Secondary Drinking Water Regulations. **Federal Register**. 56 FR 3526, January 30, 1991.

USEPA. 1991d. Drinking Water; National Primary Drinking Water Regulations; Monitoring for Volatile Organic Chemicals; MCLGs and MCLs for Aldicarb, Aldicarb Sulfoxide, Aldicarb Sulfone, Pentachlorophenol, and Barium. **Federal Register**. 56 FR 30266, July 1, 1991.

USEPA. 1992. Drinking Water; National Primary Drinking Water Regulations—Synthetic Organic Chemicals and Inorganic Chemicals; National Primary Drinking Water Regulations Implementation. **Federal Register**. 57 FR 31776, July 17, 1992.

USEPA. 1997. *Discussion Summary: EPA Technology Design Workshop.* Available on the internet at: *https://downloads.regulations.gov/EPA-HQ-OW-2018-0780-0197/content.pdf.*

USEPA. 1998a. National Primary Drinking Water Regulations: Disinfectants and Disinfection Byproducts. **Federal Register**. 63 FR 69390–69476. December 16, 1998.

USEPA. 1998b. Announcement of Small System Compliance Technology Lists for Existing National Primary Drinking Water Regulation and Findings Concerning Variance Technologies. **Federal Register**. 63 FR 42032. August 6, 1998.

USEPA. 1998c. Revision of Existing Variance and Exemption Regulations To Comply With Requirements of the Safe Drinking Water Act; Final Rule. **Federal Register**. 63 FR 43834, August 14, 1998.

USEPA. 1998d. National Primary Drinking Water Regulations: Consumer Confidence; Proposed Rule. **Federal Register**. 63 FR 7620, February 13, 1998.

USEPA. 1998e. National Primary Drinking Water Regulations: Consumer Confidence Reports. **Federal Register**. 63 FR 44524, August 19, 1998.

USEPA. 1998f. *An Assessment of the Vulnerability of Non-community water systems to SDWA cost increases.*

USEPA. 2000a. *Supplementary Guidance for Conducting Health Risk Assessment of Chemical Mixtures.* EPA 630–R–00–002. Available on the internet at: *https://cfpub.epa.gov/ncea/risk/recordisplay.cfm?deid=20533.*

USEPA. 2000b. *EPA and 3M Announce Phase Out of PFOS.* News Release. May 16, 2000. Available on the internet at: *https://archive.epa.gov/epapages/newsroom_archive/newsreleases/33aa946e6cb11f35852568e1005246b4.html.*

USEPA. 2000c. National Primary Drinking Water Regulations; Radionuclides; Final Rule. **Federal Register**. 65 FR 76708. December 7, 2000.

USEPA. 2000d. *Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health (2000).* EPA 822–B–00–004. Available on the internet at: *https://www.epa.gov/sites/default/files/2018-10/documents/methodology-wqc-protection-hh-2000.pdf.*

USEPA. 2000e. *Arsenic Occurrence in Public Drinking Water Supplies.* EPA 815–R–00–023.

USEPA. 2000f. National Primary Drinking Water Regulations; Public Notification Rule; Final Rule. **Federal Register**. 65 FR 25992. May 4, 2000.

USEPA. 2000g. *Geometries and Characteristics of Public Water Systems.* EPA–815–R–00–24.

USEPA. 2000h. *Water Supply Guidance Manual.* EPA 816–R–00–003.

USEPA. 2001. National Primary Drinking Water Regulations; Arsenic and Clarifications to Compliance and New Source Contaminants Monitoring. **Federal Register**. 66 FR 28342. July 19, 2001.

USEPA. 2002a. *A Review of the Reference Dose and Reference Concentration Process.* EPA 630–P–02–002F. Available on the internet at: *https://www.epa.gov/sites/default/files/2014-12/documents/rfd-final.pdf.*

USEPA. 2002b. *Affordability Criteria for Small Drinking Water Systems: An EPA Science Advisory Board Report.* EPA–SAB–EEAC–03–004.

USEPA. 2002c. *Implementation Guidance for the Arsenic Rule, Exemptions & the Arsenic Rule.* EPA 816–R–02–021. August 2002. Available on the internet at: *https://www.epa.gov/sites/default/files/2015-09/documents/2005_11_10_arsenic_ars_final_app_g.pdf.*

USEPA. 2002d. *Hepatocellular hypertrophy.* HED guidance document #G2002.01.

USEPA. 2003. *A Summary of General Assessment Factors for Evaluating the Quality of Scientific and Technical Information.* EPA/100/B–03/001. June 2003.

USEPA. 2004. *Drinking Water Health Advisory for Manganese.* EPA–822–R–04–003. January, 2004. Available at: *https://www.epa.gov/sites/default/files/2014-09/documents/support_cc1_magnese_dwreport_0.pdf.*

USEPA. 2005a. *Guidelines for Carcinogen Risk Assessment.* EPA/630/P–03/001F. Available on the internet at: *https://www.epa.gov/sites/default/files/2013-09/documents/cancer_guidelines_final_3-25-05.pdf.*

USEPA. 2005b. *Manual for the Certification of Laboratories Analyzing Drinking Water, Fifth Edition.* EPA 815–R–05–004. Available on the internet at *http://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=30006MXP.txt.*

USEPA 2005c. *Economic Analysis for the Final Stage 2 Disinfectants and Disinfection Byproducts Rule.* EPA 815–R–05–010.

USEPA. 2006a. *National Primary Drinking Water Regulations: Stage 2 Disinfectants and Disinfection Byproducts Rule; Final Rule.* EPA–HQ–OW–2002–0043; FRL–8012–1. *https://www.govinfo.gov/content/pkg/FR-2006-01-04/pdf/06-3.pdf.*

USEPA. 2006b. *Guidance on Systematic Planning using the Data Quality Objectives Process.* EPA/240/B–06/001. February 2006.

USEPA. 2006c. National Primary Drinking Water Regulations: Long Term 2 Enhanced Surface Water Treatment Rule. **Federal Register**. 71 FR 654. January 5, 2006.

USEPA. 2007. *Concepts, Methods and Data Sources for Cumulative Health Risk Assessment of Multiple Chemicals, Exposures and Effects: A Resource Document.* EPA/600/R–06/013F. EPA, Washington, DC. Available on the internet at: *https://cfpub.epa.gov/ncea/risk/recordisplay.cfm?deid=190187.*

USEPA. 2009a. Drinking Water Contaminant Candidate List 3-Final. **Federal Register**. 74 FR 51850. October 8, 2009.

USEPA. 2009b. *Method 537.1: Determination of Selected Per- and Polyfluorinated Alkyl Substances in Drinking Water by Solid Phase Extraction and Liquid Chromatography/Tandem Mass Spectrometry (LC/MS/MS).* EPA/600/R–18/352. Available on the internet at: *https://cfpub.epa.gov/si/si_public_record_Report.cfm?dirEntryId=343042&Lab=NERL.*

USEPA. 2009c. *Community Water System Survey, Volume II: Detailed Tables and Survey Methodology.* EPA 815–R–09–002. Available on the internet at: *http://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P1009USA.txt.*

USEPA. 2010. *Fluoride: Exposure and Relative Source Contribution Analysis.* EPA 820–R–10–015. December 2010. Available on the internet at: *https://www.epa.gov/sites/default/files/2019-03/documents/fluoride-exposure-relative-report.pdf.*

USEPA. 2011. *Recommended use of body weight 3/4 as the default method in derivation of the oral reference dose (pp. 1–50).* (EPA/100/R11/0001). Washington, DC: U.S. Environmental Protection Agency, Risk Assessment Forum, Office of the Science Advisor. *https://www.epa.gov/sites/production/files/2013-09/documents/recommended-use-of-bw34.pdf.*

USEPA. 2012. *Benchmark Dose Technical Guidance.* EPA/100/R–12/001. June 2012. Available on the internet at: *https://www.epa.gov/sites/default/files/2015-01/documents/benchmark_dose_guidance.pdf.*

USEPA. 2014a. Protocol for the Regulatory Determinations 3. Including Appendices A–F. April 2014. EPA 815–R–14–005. Available on the internet at: *https://www.regulations.gov/document/EPA-HQ-OW-2012-0217-0086.*

USEPA. 2014b. *Framework for Human Health Risk Assessment to Inform Decision Making.* EPA 100–R–14–001. April 2014. Washington, DC: U.S. Environmental Protection Agency, Office of the Science Advisor, Risk Assessment Forum.

USEPA. 2016a. Drinking Water Contaminant Candidate List 4-Final. **Federal Register**. 81 FR 81099. November 17, 2016.

USEPA. 2016b. *Six-Year Review 3—Health Effects Assessment for Existing Chemical and Radionuclide National Primary Drinking Water Regulations—Summary Report.* EPA 822–R–16–008. Available on the internet at: *https://www.epa.gov/sites/default/files/2016-12/documents/822r16008.pdf.*

USEPA. 2016c. *Health Effects Support Document for Perfluorooctanoic Acid (PFOA).* EPA 822–R–16–003. Available on the internet at: *https://www.epa.gov/sites/default/files/2016-05/documents/pfoa_hesd_final-plain.pdf.*

USEPA. 2016d. *Health Effects Support Document for Perfluorooctane Sulfonate (PFOS).* EPA 822–R–16–002. Available on the internet at: *https://www.epa.gov/sites/default/files/2016-05/documents/pfos_hesd_final_508.pdf.*

USEPA. 2016e. *Guidelines for Preparing Economic Analyses.* Available on the internet at: *https://www.epa.gov/sites/default/files/2017-08/documents/ee-0568-50.pdf.*

USEPA. 2016f. *Technical Guidance for Assessing Environmental Justice in Regulatory Analysis.* Available on the internet at: *https://www.epa.gov/sites/production/files/2016-06/documents/ejtg_5_6_16_v5.1.pdf.*

USEPA. 2017. *Third Unregulated Contaminant Monitoring Rule: Occurrence Data.* Available on the internet at: *https://www.epa.gov/dwucmr/occurrence-data-unregulated-contaminant-monitoring-rule#3.*

USEPA. 2018a. Basic Information on PFAS. Available on the internet at: *https://19january2021snapshot.epa.gov/pfas/basic-information-pfas_.html.*

USEPA. 2018b. *Response to Peer Review Comments on the Draft Human Health Toxicity Values for Hexafluoropropylene Oxide (HFPO) Dimer Acid and Its Ammonium Salt (CASRN 13252–13–6 and CASRN 62037–80–3) Also Known as ''GenX Chemicals.''* EPA 823–R–18–003. November 2018. Available on the internet at: *https://www.epa.gov/sites/default/files/2018-11/documents/genx_epa_response_to_peer_review_comments_nov2018-508.pdf.*

USEPA 2018c. Region 4 Human Health Risk Assessment Supplemental Guidance. Scientific Support Section, Superfund Division, EPA Region 4. March 2018 Update. Available on the internet at: *https://www.epa.gov/sites/default/files/2018-03/documents/hhra_regional_supplemental_guidance_report-march-2018_update.pdf.*

USEPA. 2018c. *Region 4 Human Health Risk Assessment Supplemental Guidance.* March 2018 Update. Available on the internet at: *https://www.epa.gov/sites/default/files/2018-03/documents/hhra_regional_supplemental_guidance_report-march-2018_update.pdf.*

USEPA. 2019a. *Update for Chapter 3 of the Exposure Factors Handbook: Ingestion of Water and Other Select Liquids.* EPA 600–R–18–259F. Available on the internet at: *https://www.epa.gov/sites/default/files/2019-02/documents/efh_-_chapter_3_update.pdf.*

USEPA. 2019b. *Method 533: Determination of Per- and Polyfluoroalkyl Substances in Drinking Water by Isotope Dilution Anion Exchange Solid Phase Extraction and Liquid Chromatography/Tandem Mass Spectrometry.* EPA 815–B–19–020. Available on the internet at: *https://www.epa.gov/sites/default/files/2019-12/documents/method-533-815b19020.pdf.*

USEPA. 2019c. *Health Risk Reduction and Cost Analysis of the Proposed Perchlorate National Primary Drinking Water Regulation.* EPA 816–R–19–004.

USEPA. 2019d. *Supplemental Environmental Assessment for Proposed Revisions to Effluent Limitations Guidelines and Standards for the Steam Electric Power Generating Point Source Category (EA).*

USEPA. 2019e. *EJSCREEN Technical Documentation.* Available on the internet at: *https://www.epa.gov/sites/default/files/2021-04/documents/ejscreen_technical_document.pdf.*

USEPA. 2020a. Announcement of Preliminary Regulatory Determinations for Contaminants on the Fourth Drinking Water Contaminant Candidate List. **Federal Register**. 85 FR 14098, March 10, 2020.

USEPA. 2020b. *UCMR 5 Laboratory Approval Manual Version 2.0.* December 2021. EPA 815–B–21–010. Available on the internet at: *https://www.regulations.gov/document/EPA-HQ-OW-2020-0530-0129.*

USEPA. 2020c. *Determination of Selected Per- and Polyfluorinated Alkyl Substances in Drinking Water by Solid Phase Extraction and Liquid Chromatography/Tandem Mass Spectrometry (LC/MS/MS).* EPA 600–R–20–006, Version 2.0, March 2020. Available on the internet at: *https://cfpub.epa.gov/si/si_public_record_report.cfm?Lab=NERL&dirEntryId=348508.*

USEPA. 2020d. *Interim Guidance on the Destruction and Disposal of Perfluoroalkyl and Polyfluoroalkyl Substances and Materials Containing Perfluoroalkyl and Polyfluoroalkyl Substances.* EPA–HQ–OLEM–2020–0527–0002. Available on the internet at: *https://www.epa.gov/system/files/documents/2021-11/epa-hq-olem-2020-0527-0002_content.pdf.*

USEPA. 2020e. *Labor Costs for National Drinking Water Rules.* Report prepared for EPA under Contract # EP–B16C–0001.

USEPA. 2020f. *Economic Analysis for the Final Lead and Copper Rule Revisions.* EPA 816–R–20–008.

USEPA. 2021a. *Human Health Toxicity Values for Perfluorobutane Sulfonic Acid (CASRN 375–73–5) and Related Compound Potassium Perfluorobutane Sulfonate (CASRN 29420–49–3).* EPA/600/R–20/345F. Available on the internet at: *https://cfpub.epa.gov/ncea/risk/recordisplay.cfm?deid=350888.*

USEPA. 2021b. *Human Health Toxicity Values for Hexafluoropropylene Oxide (HFPO) Dimer Acid and Its Ammonium Salt (CASRN 13252–13–6 and CASRN 62037–80–3). Also Known as ''GenX Chemicals.''* EPA/822/R–21/010. Available on the internet at: *https://www.epa.gov/system/files/documents/2021-10/genx-chemicals-toxicity-assessment_tech-edited_oct-21-508.pdf.*

USEPA. 2021c. *Assessing and Managing Chemicals under TSCA: Fact Sheet PFOA Stewardship Program.* Available on the internet at: *https://www.epa.gov/assessing-and-managing-chemicals-under-tsca/fact-sheet-20102015-pfoa-stewardship-program.*

USEPA. 2021d. Announcement of Final Regulatory Determinations for Contaminants on the Fourth Drinking Water Contaminant Candidate List. **Federal Register**. 86 FR 12272, March 3, 2021.

USEPA. 2021e. *Draft Framework for Estimating Noncancer Health Risks Associated with Mixtures of Per- and Polyfluoroalkyl Substances (PFAS).* EPA–822–D–21–003. Available on the internet at: *https://sab.epa.gov/ords/sab/f?p=100:18:10311539418988:::18:P18_ID:2601.*

USEPA. 2021f. *Response To Additional Focused External Peer Review of Draft Human Health Toxicity Values for Hexafluoropropylene Oxide (HFPO) Dimer Acid and Its Ammonium Salt (GenX Chemicals).* Available on the internet at: *https://www.epa.gov/system/files/documents/2021-10/epa_2nd-response-to-peer-review_genx_508.pdf.*

USEPA. 2021g. *EPA Response to Public Comments on Draft Human Health Toxicity Values for Hexafluoropropylene Oxide (HFPO) Dimer Acid and Its Ammonium Salt (CASRN 13252–13–6 and CASRN 62037–80–3) Also Known as ''GenX Chemicals''.* EPA 822R–21–008. October 2021. Available on the internet at: *https://www.epa.gov/system/files/documents/2021-10/final-genx-assessment-resp-to-public-comments_508.pdf.*

USEPA. 2021h. *Systematic Review Protocol for the PFAS IRIS Assessments.* EPA/635/R–19/050, 2019. Available on the internet at: *https://cfpub.epa.gov/ncea/iris_drafts/recordisplay.cfm?deid=345065#tab-3.*

USEPA. 2021i. *Proposed Approaches to the Derivation of a Draft Maximum Contaminant Level Goal for Perfluorooctanoic Acid (PFOA) (CASRN 335–67–1) in Drinking Water.* EPA 822–D–21–001. Available on the internet at: *https://sab.epa.gov/ords/sab/f?p=100:18:16490947993:::RP,18:P18_ID:2601.*

USEPA. 2021j. *Proposed Approaches to the Derivation of a Draft Maximum Contaminant Level Goal for Perfluorooctane Sulfonic Acid (PFOS) (CASRN 1763–23–1) in Drinking Water.* EPA 822–D–21–002. Available on the internet at: *https://sab.epa.gov/ords/sab/f?p=100:18:16490947993:::RP,18:P18_ID:2601.*

USEPA. 2021k. *Analysis of Cardiovascular Disease Risk Reduction as a Result of Reduced PFOA and PFOS Exposure in Drinking Water.* EPA 815–D–21–001. Available on the internet at: *https://sab.epa.gov/ords/sab/f?p=100:18:16490947993:::RP,18:P18_ID:2601.*

USEPA. 2022a. Changes to Reporting Requirements for Per- and Polyfluoroalkyl Substances and to Supplier Notifications for Chemicals of Special Concern; Community Right-to-Know Toxic Chemical Release Reporting. **Federal Register**. 87 FR 74379, December 5, 2022.

USEPA. 2022b. Drinking Water Contaminant Candidate List 5—Final. **Federal Register**. 87 FR 68060, November 14, 2022.

USEPA. 2022c. *PFAS Strategic Roadmap: EPA's Commitments to Action 2021–2024.* Available on the internet at: *https://www.epa.gov/pfas/pfas-strategic-roadmap-epas-commitments-action-2021-2024.*

USEPA. 2022d. *Addressing PFAS discharges in EPA-issued NPDES permits and expectations where EPA is the pretreatment control authority (April 28, 2022). https://www.epa.gov/system/files/documents/2022-04/npdes_pfas-memo.pdf.*

USEPA. 2022e. *Addressing PFAS discharges in NPDES permits and through the pretreatment program and monitoring programs* (December 5, 2022). *https://www.epa.gov/system/files/documents/2022-12/NPDES_PFAS_State%20Memo_December_2022.pdf.*

USEPA. 2022f. *ORD Staff Handbook for Developing IRIS Assessments.* EPA 600/R–22/268. Available on the internet at: *https://cfpub.epa.gov/ncea/iris_drafts/recordisplay.cfm?deid=356370.*

USEPA. 2022g. *IRIS Toxicological Review of Perfluorobutanoic Acid (PFBA) and Related Salts (Final Report, 2022).* EPA/635/R–22/277F. Retrieved from *https://www.epa.gov/chemical-research/iris-toxicological-review-perfluorobutanoic-acid-pfba-and-related-salts-final.*

USEPA. 2022h. *Letter is in response to the Request for Correction (RFC) received by the USEPA from Arnold & Porter Kaye Scholer LLP (A&P) on March 18, 2022.* Available on the internet at: *https://www.epa.gov/system/files/documents/2022-06/RFC_22001_Response_June2022.pdf.*

USEPA. 2022i. *Transmittal of the Science Advisory Board Report titled, ''Review of EPA's Analyses to Support EPA's National Primary Drinking Water Rulemaking for PFAS.''* EPA–22–008. Available on the internet at: *https://sab.epa.gov/ords/sab/f?p=114:12:15255596377846.*

USEPA. 20222j. Fifth Unregulated Contaminant Monitoring Rule. Available on the internet at: *https://www.epa.gov/dwucmr/fifth-unregulated-contaminant-monitoring-rule.*

USEPA. 2022k. MRL Data of UCMR 5 Approved Labs.

USEPA. 2022l. *Comprehensive Environmental Response, Compensation, and Liability Act Hazardous Substances: Designation of Perfluorooctanoic Acid and Perfluorooctanesulfonic Acid.* EPA–HQ–OLEM–2019–0341–0001. Washington, DC. Retrieved from *https://*

www.regulations.gov/document/EPA-HQ-OLEM-2019-0341-0001.

USEPA. 2022m. *National Overview: Facts and Figures on Materials, Wastes and Recycling (updated: December 3, 2022). https://www.epa.gov/facts-and-figures-about-materials-waste-and-recycling/national-overview-facts-and-figures-materials.*

USEPA. 2023a. *Emerging Contaminants (EC) in Small or Disadvantaged Communities Grant (SDC).* Available on the internet at: *https://www.epa.gov/dwcapacity/emerging-contaminants-ec-small-or-disadvantaged-communities-grant-sdc.*

USEPA. 2023b. *Water Technical Assistance (WaterTA).* Available on the internet at: *https://www.epa.gov/water-infrastructure/water-technical-assistance-waterta.*

USEPA. 2023c. Advances in Dose Addition for Chemical Mixtures: A White Paper. U.S. Environmental Protection Agency, Washington, DC. EPA/100/R23/001.

USEPA. 2023d. *IRIS Toxicological Review of Perfluorohexanesulfonic Acid (PFHxS) and Related Salts (Public Comment and External Review Draft).* EPA/635/R–23/056. Washington, DC.

USEPA. 2023e. *Public Comment Draft—Maximum Contaminant Level Goal (MCLG) Summary Document for a Mixture of Four Per- and Polyfluoroalkyl Substances (PFAS): GenX Chemicals, PFBS, PFNA and PFHxS.* EPA–822–P–23–004.

USEPA. 2023f. PFAS National Primary Drinking Water Regulation Rulemaking. **Federal Register**. 88 FR 18638. March 29, 2023.

USEPA. 2023g. *Public Comment Draft—Toxicity Assessment and Proposed Maximum Contaminant Level Goal (MCLG) for Perfluorooctanoic Acid (PFOA) (CASRN 335–67–1) in Drinking Water.* EPA–822–P–23–005.

USEPA. 2023h. *Public Comment Draft—Toxicity Assessment and Proposed Maximum Contaminant Level Goal (MCLG) for Perfluorooctane Sulfonic Acid (PFOS) (CASRN 1763–23–1) in Drinking Water.* EPA–822–P–23–007.

USEPA. 2023i. *Public Comment Draft—Appendix: Proposed Maximum Contaminant Level Goal for Perfluorooctanoic Acid (PFOA) (CASRN 335–67–1) in Drinking Water.*

USEPA. 2023j. *Public Comment Draft—Appendix: Proposed Maximum Contaminant Level Goal for Perfluorooctane Sulfonic Acid (PFOS) (CASRN 1763–23–1) in Drinking Water.*

USEPA. 2023k. *EPA Response to Final Science Advisory Board Recommendations (August 2022) on Four Draft Support Documents for the EPA's Proposed PFAS National Primary Drinking Water Regulation.* EPA–822–D–23–001.

USEPA. 2023l. *PFAS Occurrence and Contaminant Background Support Document.* EPA–822–P–23–010.

USEPA. 2023m. *EPA Response to Letter of Peer Review for Disinfectant Byproduct Reduction as a Result of Granular Activated Carbon Treatment for PFOA and PFOS in Drinking Water: Benefits Analysis Related to Bladder Cancer.* EPA–815–B23–001.

USEPA. 2023n. *Economic Analysis of the Proposed National Primary Drinking Water Regulation for Per- and Polyfluoroalkyl Substances.* EPA–822–P–23–001.

USEPA. 2023o. *Appendix: Economic Analysis of the Proposed National Primary Drinking Water Regulation for Per- and Polyfluoroalkyl Substances.* EPA–822–P–23–002.

USEPA. 2023p. *IRIS Toxicological Review of Perfluorohexanoic Acid [PFHxA, CASRN 307–24–4] and Related Salts* (EPA/635/R–23/027Fa). Retrieved from *https://cfpub.epa.gov/ncea/iris/iris_documents/documents/toxreviews/0704tr.pdf.*

USEPA. 2023q. Federal ''Good Neighbor Plan'' for the 2015 Ozone National Ambient Air Quality Standards. **Federal Register**. 88 FR 36654. June 5, 2023.

USEPA. 2023r. New Source Performance Standards for Greenhouse Gas Emissions From New, Modified, and Reconstructed Fossil Fuel-Fired Electric Generating Units; Emission Guidelines for Greenhouse Gas Emissions From Existing Fossil Fuel-Fired Electric Generating Units; and Repeal of the Affordable Clean Energy Rule. **Federal Register**. 88 FR 33240. May 23, 2023.

USEPA. 2023s. *Regulatory Impact Analysis of the Standards of Performance for New, Reconstructed, and Modified Sources and Emissions Guidelines for Existing Sources: Oil and Natural Gas Sector Climate Review.* EPA–452/R–23–013.

USEPA. 2023t. Supplementary Material for the Regulatory Impact Analysis for the Final Rulemaking, ''Standards of Performance for New, Reconstructed, and Modified Sources and Emissions Guidelines for Existing Sources: Oil and Natural Gas Sector Climate Review''; EPA Report on the Social Cost of Greenhouse Gases: Estimates Incorporating Recent Scientific Advances.

USEPA. 2023u. New Source Performance Standards for the Synthetic Organic Chemical Manufacturing Industry and National Emission Standards for Hazardous Air Pollutants for the Synthetic Organic Chemical Manufacturing Industry and Group I & II Polymers and Resins Industry. **Federal Register**. 88 FR 25080. April 25, 2023.

USEPA. 2023v. Executive Order 14096—Revitalizing Our Nation's Commitment to Environmental Justice for All. **Federal Register**. 88 FR 25251, April 26, 2023.

USEPA. 2023w. Framework for Estimating Noncancer Health Risks Associated with Mixtures of Per- and Polyfluoroalkyl Substances (PFAS). EPA–822–P–23–003.

USEPA. 2024a. *Framework for Estimating Noncancer Health Risks Associated with Mixtures of Per- and Polyfluoroalkyl Substances (PFAS).* 815–R–24–003.

USEPA. 2024b. *PFAS Occurrence and Contaminant Background Support Document for the Final PFAS NPDWR.* 815–R–24–013.

USEPA. 2024c. *Office of Water Final Human Health Toxicity Assessment for Perfluorooctanoic Acid (PFOA).* 815–R–24–006.

USEPA 2024d. *Office of Water Final Human Health Toxicity Assessment for Perfluorooctane Sulfonic Acid (PFOS).* 815–R–24–007.

USEPA. 2024e. *Economic Analysis for the Final Per- and Polyfluoroalkyl Substances National Primary Drinking Water Regulation Appendices.* 815–R–24–002.

USEPA 2024f. *Maximum Contaminant Level Goals (MCLGs) for Three Individual Per- and Polyfluoroalkyl Substances (PFAS) and a Mixture of Four PFAS.* 815–R–24–004.

USEPA. 2024g. *Economic Analysis for the Final Per- and Polyfluoroalkyl Substances National Primary Drinking Water Regulation.* 815–R–24–001.

USEPA. 2024h. *Appendix—Office of Water Final Human Health Toxicity Assessment for Perfluorooctanoic Acid (PFOA).* 815–R–24–008.

USEPA. 2024i. *Appendix—Office of Water Final Human Health Toxicity Assessment for Perfluorooctane Sulfonic Acid (PFOS).* 815–R–24–009.

USEPA. 2024j. *OW Final Maximum Contaminant Level Goals (MCLGs) for Perfluorooctanoic Acid (PFOA) and Perfluorooctane Sulfonic Acid (PFOS) in Drinking Water.* 815–R–24–010.

USEPA. 2024k. *EPA's Responses to Public Comments on the Proposed PFAS National Primary Drinking Water Regulation.* 815–R–24–005.

USEPA. 2024l. *Best Available Technologies and Small System Compliance Technologies Per- and Polyfluoroalkyl Substances (PFAS) in Drinking Water.* 815–R–24–011.

USEPA. 2024m. *Technologies and Costs for Removing Per- and Polyfluoroalkyl Substances from Drinking Water.* 815–R–24–012.

USEPA. 2024n. *Fifth Unregulated Contaminant Monitoring Rule: Occurrence Data.* Available on the internet at: *https://www.epa.gov/dwucmr/occurrence-data-unregulated-contaminant-monitoring-rule#5.*

USEPA. 2024o. Fluoride Informational Links. Available on the internet at: *https://tdb.epa.gov/tdb/contaminant?id=10700.*

USEPA. 2024p. *Work Breakdown Structure-Based Cost Model for Granular Activated Carbon Drinking Water Treatment.*

USEPA. 2024q. *Work Breakdown Structure-Based Cost Model for Ion Exchange Treatment of Per- and Polyfluoroalkyl Substances (PFAS) in Drinking Water.*

USEPA. 2024r. *Work Breakdown Structure-Based Cost Model for Nontreatment Options for Drinking Water Compliance.*

United States Office of Science and Technology Policy (OSTP). 2023. *Per- and Polyfluoroalkyl Substances (PFAS) Report,* A Report by the Joint Subcommittee on Environment, Innovation, and Public Health. Available on the internet at: *https://www.whitehouse.gov/wpcontent/uploads/2023/03/OSTP-March-2023-PFASReport.pdf.*

Valvi, D., Oulhote, Y., Weihe, P., Dalgård, C., Bjerve, K.S., Steuerwald, U., and

Grandjean, P. 2017. Gestational Diabetes and Offspring Birth Size at Elevated Environmental Pollutant Exposures. *Environment International,* 107:205–215. *https://doi.org/10.1016/j.envint.2017.07.016.*

van de Schoot, R., Depaoli, S., King, R., Kramer, B., Märtens, K., Tadesse, M.G., Vannucci, M., Gelman, A., Veen, D., Willemsen, J., and Yau, C. 2021. Bayesian statistics and modelling. *Nature Reviews Methods Primers, 1*(1):1. *https://doi.org/10.1038/s43586-020-00003-0.*

Vélez, M., Arbuckle, T., and Fraser, W. 2015. Maternal exposure to perfluorinated chemicals and reduced fecundity: the MIREC study. *Human Reproduction,* 30(3):701–709. *https://doi.org/10.1093/humrep/deu350.*

Verner, M.A., Loccisano, A.E., Morken, N.H., Yoon, M., Wu, H., McDougall, R., Maisonet, M., Marcus, M., Kishi, R., Miyashita, C., Chen., M.-H, Hsieh, W.-S, Andersen, M.E., Clewell III, H.J., and Longnecker, M.P. 2015. Associations of Perfluoroalkyl Substances (PFAS) with Lower Birth Weight: An Evaluation of Potential Confounding by Glomerular Filtration Rate Using a Physiologically Based Pharmacokinetic Model (PBPK). *Environmental Health Perspectives,* 123(12):1317–1324. *https://doi.org/10.1289/ehp.1408837.*

Vieira, V.M., Hoffman, K., Shin, H., Weinberg, J.M., Webster, T.F., and Fletcher, T. 2013. Perfluorooctanoic Acid Exposure and Cancer Outcomes in a Contaminated Community: A Geographic Analysis. *Environmental Health Perspectives,* 121(3):318–323. *https://doi.org/10.1289/ehp.1205829.*

Villanueva, C.M., Cantor, K.P., Cordier, S., Jaakkola, J.J.K., King, W.D., Lynch, C.F., Porru, S., and Kogevinas, M. 2004. Disinfection Byproducts and Bladder Cancer: a Pooled Analysis. *Epidemiology,* 15(3):357–367. *https://doi.org/10.1097/01.ede.0000121380.02594.fc.*

Villanueva, C.M., Cantor, K.P., King, W.D., Jaakkola, J.J., Cordier, S., Lynch, C.F., Porru, S., and Kogevinas, M. 2006. Total and Specific Fluid Consumption as Determinants of Bladder Cancer Risk. *International Journal of Cancer,* 118(8):2040–2047. *https://doi.org/10.1002/ijc.21587.*

Villanueva, C.M., Cantor, K.P., Grimalt, J.O., Malats, N., Silverman, D., Tardon, A., Garcia-Closas, R., Serra, C., Carrato., A., Castano-Vinyals, G., Marcos, R., Rothman, N., Real, F.X., Dosemeci, M., and Kogevinas, M. 2007. Bladder Cancer and Exposure to Water Disinfection By-products through Ingestion, Bathing, Showering, and Swimming in Pools. *American Journal of Epidemiology,* 156(2):148–156. *https://doi.org/10.1093/aje/kwj364.*

Wang, Y., Adgent, M., Su, P.-H., Chen, H.-Y., Chen, P.-C., Hsiung, C.A., and Wang, S.-L. 2016. Prenatal exposure to perfluorocarboxylic acids (PFCAs) and fetal and postnatal growth in the Taiwan Maternal and Infant Cohort Study. *Environmental Health Perspectives,* 124(11):1794–1800.

Wang, L., Vacs Renwick, D., and Regli, S. 2019. Re-assessing effects of bromide and granular activated carbon on disinfection byproduct formation. *AWWA Water Science,* 1(4). *https://doi.org/10.1002/aws2.1147.*

Waterfield, G., Rogers, M., Grandjean, P., Auffhammer, M., and Sunding, D. 2020. Reducing Exposure to High Levels of Perfluorinated Compounds in Drinking Water Improves Reproductive Outcomes: Evidence from an Intervention in Minnesota. *Environmental Health,* 19:1–11. *https://doi.org/10.1186/s12940-020-00591-0.*

Weisman, R.J., Heinrich, A., Letkiewicz, F., Messner, M., Studer, K., Wang, L., and Regli, S. 2022. Estimating National Exposures and Potential Bladder Cancer Cases Associated with Chlorination DBPs in US Drinking Water. *Environmental Health Perspectives,* 130(8):087002. *https://doi.org/10.1289/EHP9985.*

Wikström, S., Lin, P.I., Lindh, C.H., Shu, H., and Bornehag, C.G. 2020. Maternal Serum Levels of Perfluoroalkyl Substances in Early Pregnancy and Offspring Birth Weight. *Pediatric Research,* 87(6):1093–1099. *https://doi.org/10.1038/s41390-019-0720-1.*

Windham, G., and Fenster, L. 2008. Environmental Contaminants and pregnancy Outcomes. *Fertility and Sterility,* 89(2):e111–e116. *https://doi.org/10.1016/j.fertnstert.2007.12.041.*

Wright, J.M., Lee, A.L., Rappazzo, K., Ru, H., Radke, E., and Bateson, T.F. 2023. Systematic Review and Meta-analysis of Birth Weight and PFNA Exposures. *Environmental Research,* 222:115357. doi: 10.1016/j.envres.2023.115357. Epub 2023 Jan 24. PMID: 36706898.

Yao, Q., Gao, Y., Zhang, Y., Qin, K., Liew, Z., and Tian, Y. 2021. Associations of paternal and maternal per- and polyfluoroalkyl substances exposure with cord serum reproductive hormones, placental steroidogenic enzyme and birth weight. *Chemosphere,* 285:131521.

Yapsakli, K., and Çeçen, F. 2010. Effect of Type of Granular Activated Carbon on DOC Biodegradation in Biological Activated Carbon Filters. *Process Biochemistry,* 45(3):355–362. *https://doi.org/10.1016/j.procbio.2009.10.005.*

Yin, X., Di, T., Cao, X., Liu, Z., Xie, J., and Zhang, S. 2021. Chronic exposure to perfluorohexane sulfonate leads to a reproduction deficit by suppressing hypothalamic kisspeptin expression in mice. *Journal of Ovarian Research,* 14:1–13. *https://doi.org/10.1186/s13048-021-00903-z.*

Zaggia, A., Conte, L., Falletti, L., Fant, M., and Chiorboli, A. 2016. Use of strong anion exchange resins for the removal of perfluoroalkylated substances from contaminated drinking water in batch and continuous pilot plants. *Water Research,* 91:137–146. *https://doi.org/10.1016/j.watres.2015.12.039.*

List of Subjects

40 CFR Part 141

Environmental protection, Incorporation by reference, Indians—lands, Intergovernmental relations, Monitoring and analytical requirements, Per- and polyfluoroalkyl substances, Reporting and recordkeeping requirements, Water supply.

40 CFR Part 142

Environmental protection, Administrative practice and procedure, Indians—lands, Intergovernmental relations, Monitoring and analytical requirements, Per- and polyfluoroalkyl substances, Reporting and recordkeeping requirements, Water supply.

Michael S. Regan,
Administrator.

For the reasons stated in the preamble, the Environmental Protection Agency amends 40 CFR parts 141 and 142 as follows:

PART 141—NATIONAL PRIMARY DRINKING WATER REGULATIONS

■ 1. The authority citation for part 141 continues to read as follows:

Authority: 42 U.S.C. 300f, 300g–1, 300g–2, 300g–3, 300g–4, 300g–5, 300g–6, 300j–4, 300j–9, and 300j–11.

■ 2. Amend § 141.2 by adding in alphabetical order the definitions for ‘‘Hazard Index (HI)’’, ‘‘Hazard quotient (HQ)’’, ‘‘Health-based water concentration (HBWC)’’, ‘‘HFPO–DA or GenX chemicals’’, ‘‘PFBS’’, ‘‘PFHxS’’, ‘‘PFNA’’, ‘‘PFOA’’, and ‘‘PFOS’’ to read as follows:

§ 141.2 Definitions.

* * * * *

Hazard Index (HI) is the sum of component hazard quotients (HQs), which are calculated by dividing the measured regulated PFAS component contaminant concentration in water (*e.g.,* expressed as parts per trillion (ppt) or nanograms per liter (ng/l)) by the associated health-based water concentration (HBWC) expressed in the same units as the measured concentration (*e.g.,* ppt or ng/l). For PFAS, a mixture Hazard Index greater than 1 (unitless) is an exceedance of the MCL.

Hazard quotient (HQ) means the ratio of the measured concentration in drinking water to the health-based water concentration (HBWC).

Health-based water concentration (HBWC) means level below which there are no known or anticipated adverse health effects over a lifetime of exposure, including sensitive populations and life stages, and allows for an adequate margin of safety.

HFPO–DA or *GenX chemicals* means Chemical Abstract Service registration number 122499–17–6, chemical formula C6F11O3-, International Union of Pure and Applied Chemistry preferred name 2,3,3,3-tetrafluoro-2-(heptafluoropropoxy)propanoate, along with its conjugate acid and any salts, derivatives, isomers, or combinations thereof.

* * * * *

PFBS means Chemical Abstract Service registration number 45187–15–3, chemical formula C4F9SO3-, perfluorobutane sulfonate, along with its conjugate acid and any salts, derivatives, isomers, or combinations thereof.

PFHxS means Chemical Abstract Service registration number 108427–53–8, chemical formula C6F13SO3-, perfluorohexane sulfonate, along with its conjugate acid and any salts, derivatives, isomers, or combinations thereof.

PFNA means Chemical Abstract Service registration number 72007–68–2, chemical formula C9F17O2-, perfluorononanoate, along with its conjugate acid and any salts, derivatives, isomers, or combinations thereof.

PFOA means Chemical Abstract Service registration number 45285–51–6, chemical formula C8F15O2-, perfluorooctanoate, along with its conjugate acid and any salts, derivatives, isomers, or combinations thereof.

PFOS means Chemical Abstract Service registration number 45298–90–6, chemical formula C8F17SO3–, perfluorooctanesulfonate, along with its conjugate acid and any salts, derivatives, isomers, or combinations thereof.

* * * * *

■ 3. Amend § 141.6 by revising paragraph (a) and adding paragraph (l) to read as follows:

§ 141.6 Effective dates.

(a) Except as provided in paragraphs (b) through (l) of this section the regulations set forth in this part take effect on June 24, 1977.

* * * * *

(l) The regulations pertaining to the per- and polyfluoroalkyl substances (PFAS) chemicals set forth in subpart Z of this part are effective June 25, 2024. See § 141.900 for the compliance dates for provisions under subpart Z. Compliance with reporting requirements under subpart Z, in accordance with subparts O (the consumer confidence rule) and Q (the public notification rule) of this part are required on April 26, 2027, except for notification requirements in § 141.203 related to violations of the MCLs. The compliance date for the PFAS MCLs in § 141.61, as specified in § 141.60, and for § 141.203 notifications of violations of the PFAS MCLs is April 26, 2029.

■ 4. Amend § 141.24 by revising paragraph (h) introductory text to read as follows:

§ 141.24 Organic chemicals, sampling and analytical requirements.

* * * * *

(h) Analysis of the contaminants listed in § 141.61(c) for the purposes of determining compliance with the maximum contaminant level shall be conducted as follows, with the exceptions that this paragraph (h) does not apply to regulated PFAS (see § 141.902) and no monitoring is required for aldicarb, aldicarb sulfoxide, or aldicarb sulfone:

* * * * *

■ 5. Amend § 141.28 by revising paragraph (a) to read as follows:

§ 141.28 Certified laboratories.

(a) For the purpose of determining compliance with §§ 141.21 through 141.27, 141.40, 141.74, 141.89, 141.402, 141.901, and 141.902, samples may be considered only if they have been analyzed by a laboratory certified by EPA or the State except that measurements of alkalinity, disinfectant residual, orthophosphate, pH, silica, temperature, and turbidity may be performed by any person acceptable to the State.

* * * * *

■ 6. Amend § 141.50 by:

■ a. Adding periods at the ends of paragraphs (a)(1) through (23);

■ b. Adding paragraphs (a)(24) and (25); and

■ c. In the table to paragraph (b), revising the heading for the second column and adding in numerical order the entries ''(34),'' ''(35),'' ''(36),'' and ''(37)'' and footnote 1.

The additions and revision read as follows:

§ 141.50 Maximum contaminant level goals for organic contaminants.

(a) * * *

(24) PFOA.

(25) PFOS.

(b) * * *

Contaminant	MCLG in mg/l (unless otherwise noted)
* * * * * *	*
(34) Hazard Index PFAS (HFPO–DA, PFBS, PFHxS, and PFNA)	1 (unitless).[1]
(35) HFPO–DA	0.00001.
(36) PFHxS	0.00001.
(37) PFNA	0.00001.

[1] The PFAS Mixture Hazard Index (HI) is the sum of component hazard quotients (HQs), which are calculated by dividing the measured component PFAS concentration in water by the corresponding contaminant's health-based water concentration (HBWC) when expressed in the same units (shown in ng/l). The HBWC for PFHxS is 10 ng/l; the HBWC for HFPO–DA is 10 ng/l; the HBWC for PFNA is 10 ng/l; and the HBWC for PFBS is 2000 ng/l. A PFAS Mixture Hazard Index greater than 1 (unitless) indicates an exceedance of the health protective level and indicates potential human health risk from the PFAS mixture in drinking water.

Hazard Index = $([\text{HFPO–DA}_{\text{water}}\ \text{ng/l}]/[10\ \text{ng/l}]) + ([\text{PFBS}_{\text{water}}\ \text{ng/l}]/[2000\ \text{ng/l}]) + ([\text{PFNA}_{\text{water}}\ \text{ng/l}]/[10\ \text{ng/l}]) + ([\text{PFHxS}_{\text{water}}\ \text{ng/l}]/[10\ \text{ng/l}])$

HBWC = health-based water concentration
HQ = hazard quotient
ng/l = nanograms per liter
$\text{PFAS}_{\text{water}}$ = the concentration of a specific PFAS in water

■ 7. Amend § 141.60 by adding paragraph (a)(4) to read as follows:

§ 141.60 Effective dates.

(a) * * *

(4) The effective date for paragraphs (c)(34) through (40) of § 141.61 (listed in table 4 to paragraph (c)) is April 26, 2029.

* * * * *

■ 8. Amend § 141.61 by:

■ a. In paragraph (a), revising the introductory text and adding a table heading;

■ b. In paragraph (b), revising the introductory text and the table heading;

■ c. Revising and republishing paragraph (c); and

■ d. Adding paragraphs (d) and (e).

The revisions and additions read as follows:

§ 141.61 Maximum contaminant levels for organic contaminants.

(a) The following maximum contaminant levels for volatile organic contaminants apply to community and non-transient, non-community water systems.

TABLE 1 TO PARAGRAPH (a)—MAXIMUM CONTAMINANT LEVELS FOR VOLATILE ORGANIC CONTAMINANTS

* * * * *

(b) The Administrator, pursuant to section 1412 of the Act, hereby identifies as indicated in table 2 to this paragraph (b) granular activated carbon (GAC), packed tower aeration (PTA), or oxidation (OX) as the best technology, treatment technique, or other means available for achieving compliance with the maximum contaminant level for organic contaminants identified in paragraphs (a) and (c) of this section, except for per- and polyfluoroalkyl substances (PFAS).

TABLE 2 TO PARAGRAPH (b)—BAT FOR ORGANIC CONTAMINANTS IN PARAGRAPHS (a) AND (c) OF THIS SECTION, EXCEPT FOR PFAS

* * * * *

(c) The following maximum contaminant levels (MCLs) in tables 3 and 4 to this paragraph (c) for synthetic organic contaminants apply to community water systems and non-transient, non-community water systems; table 4 also contains health-based water concentrations (HBWCs) for selected per- and poly-fluoroalkyl substances (PFAS) used in calculating the Hazard Index.

TABLE 3 TO PARAGRAPH (c)—MCLS FOR SYNTHETIC ORGANIC CONTAMINANTS, EXCEPT FOR PFAS

CAS No.	Contaminant	MCL (mg/l)
(1) 15972–60–8	Alachlor	0.002
(2) 116–06–3	Aldicarb	0.003
(3) 1646–87–3	Aldicarb sulfoxide	0.004
(4) 1646–87–4	Aldicarb sulfone	0.002
(5) 1912–24–9	Atrazine	0.003
(6) 1563–66–2	Carbofuran	0.04
(7) 57–74–9	Chlordane	0.002
(8) 96–12–8	Dibromochloropropane	0.0002
(9) 94–75–7	2,4–D	0.07
(10) 106–93–4	Ethylene dibromide	0.00005
(11) 76–44–8	Heptachlor	0.0004
(12) 1024–57–3	Heptachlor epoxide	0.0002
(13) 58–89–9	Lindane	0.0002
(14) 72–43–5	Methoxychlor	0.04
(15) 1336–36–3	Polychlorinated biphenyls	0.0005
(16) 87–86–5	Pentachlorophenol	0.001
(17) 8001–35–2	Toxaphene	0.003
(18) 93–72–1	2,4,5–TP	0.05
(19) 50–32–8	Benzo[a]pyrene	0.0002
(20) 75–99–0	Dalapon	0.2
(21) 103–23–1	Di(2-ethylhexyl) adipate	0.4
(22) 117–81–7	Di(2-ethylhexyl) phthalate	0.006
(23) 88–85–7	Dinoseb	0.007
(24) 85–00–7	Diquat	0.02
(25) 145–73–3	Endothall	0.1
(26) 72–20–8	Endrin	0.002
(27) 1071–53–6	Glyphosate	0.7
(28) 118–74–1	Hexacholorbenzene	0.001
(29) 77–47–4	Hexachlorocyclopentadiene	0.05
(30) 23135–22–0	Oxamyl (Vydate)	0.2
(31) 1918–02–1	Picloram	0.5
(32) 122–34–9	Simazine	0.004
(33) 1746–01–6	2,3,7,8–TCDD (Dioxin)	3×10^{-8}

TABLE 4 TO PARAGRAPH (c)—MCLS AND HBWCS FOR PFAS

CAS. No.	Contaminant	MCL (mg/l) (unless otherwise noted)	HBWC (mg/l) for hazard index calculation
(34) Not applicable	Hazard Index PFAS (HFPO–DA, PFBS, PFHxS, and PFNA).	1 (unitless)[1]	Not applicable
(35) 122499–17–6	HFPO–DA	0.00001	0.00001
(36) 45187–15–3	PFBS	No individual MCL	0.002
(37) 108427–53–8	PFHxS	0.00001	0.00001
(38) 72007–68–2	PFNA	0.00001	0.00001
(39) 45285–51–6	PFOA	0.0000040	Not applicable

TABLE 4 TO PARAGRAPH (c)—MCLS AND HBWCS FOR PFAS—Continued

CAS. No.	Contaminant	MCL (mg/l) (unless otherwise noted)	HBWC (mg/l) for hazard index calculation
(40) 45298–90–6	PFOS	0.0000040	Not applicable

[1] The PFAS Mixture Hazard Index (HI) is the sum of component hazard quotients (HQs), which are calculated by dividing the measured component PFAS concentration in water by the relevant health-based water concentration when expressed in the same units (shown in ng/l for simplification). The HBWC for PFHxS is 10 ng/l; the HBWC for HFPO–DA is 10 ng/l; the HBWC for PFNA is 10 ng/l; and the HBWC for PFBS is 2000 ng/l.

$$\text{Hazard Index} = ([\text{HFPO–DA}_{\text{water}}\ \text{ng/l}]/[10\ \text{ng/l}]) + ([\text{PFBS}_{\text{water}}\ \text{ng/l}]/[2000\ \text{ng/l}]) + ([\text{PFNA}_{\text{water}}\ \text{ng/l}]/[10\ \text{ng/l}]) + ([\text{PFHxS}_{\text{water}}\ \text{ng/l}]/[10\ \text{ng/l}])$$

HBWC = health-based water concentration
HQ = hazard quotient
ng/l = nanograms per liter
PFAS$_{\text{water}}$ = the concentration of a specific PFAS in water

(d) The Administrator, pursuant to section 1412 of the Act, hereby identifies in table 5 to this paragraph (d) the best technology, treatment technique, or other means available for achieving compliance with the maximum contaminant levels for all regulated PFAS identified in paragraph (c) of this section:

TABLE 5 TO PARAGRAPH (d)—BEST AVAILABLE TECHNOLOGIES FOR PFAS LISTED IN PARAGRAPH (c) OF THIS SECTION

Contaminant	BAT
Hazard Index PFAS (HFPO–DA, PFBS, PFHxS, and PFNA)	Anion exchange, GAC, reverse osmosis, nanofiltration.
HFPO–DA	Anion exchange, GAC, reverse osmosis, nanofiltration.
PFHxS	Anion exchange, GAC, reverse osmosis, nanofiltration.
PFNA	Anion exchange, GAC, reverse osmosis, nanofiltration.
PFOA	Anion exchange, GAC, reverse osmosis, nanofiltration.
PFOS	Anion exchange, GAC, reverse osmosis, nanofiltration.

(e) The Administrator, pursuant to section 1412 of the Act, hereby identifies in table 6 to this paragraph (e) the affordable technology, treatment technique, or other means available to systems serving 10,000 persons or fewer for achieving compliance with the maximum contaminant levels for all regulated PFAS identified in paragraph (c) of this section:

TABLE 6 TO PARAGRAPH (e)—SMALL SYSTEM COMPLIANCE TECHNOLOGIES (SSCTS) FOR PFAS

Small system compliance technology [1]	Affordable for listed small system categories [2]
Granular Activated Carbon.	All size categories.
Anion Exchange	All size categories.
Reverse Osmosis, Nanofiltration [3].	3,301–10,000.

[1] Section 1412(b)(4)(E)(ii) of SDWA specifies that SSCTs must be affordable and technically feasible for small systems.

[2] The Act (ibid.) specifies three categories of small systems: (i) those serving 25 or more, but fewer than 501, (ii) those serving more than 500, but fewer than 3,301, and (iii) those serving more than 3,300, but fewer than 10,001.

[3] Technologies reject a large volume of water and may not be appropriate for areas where water quantity may be an issue.

■ 9. Amend § 141.151 by revising paragraph (d) to read as follows:

§ 141.151 Purpose and applicability of this subpart.

* * * * *

(d) For the purpose of this subpart, detected means: at or above the levels prescribed by § 141.23(a)(4) for inorganic contaminants, at or above the levels prescribed by § 141.24(f)(7) for the contaminants listed in § 141.61(a), at or above the levels prescribed by § 141.24(h)(18) for the contaminants listed in § 141.61(c) (except PFAS), at or above the levels prescribed by § 141.131(b)(2)(iv) for the contaminants or contaminant groups listed in § 141.64, at or above the levels prescribed by § 141.25(c) for radioactive contaminants, and at or above the levels prescribed in § 141.902(a)(5) for PFAS listed in § 141.61(c).

* * * * *

■ 10. Amend § 141.153 by adding paragraph (c)(3)(v) to read as follows:

§ 141.153 Content of the reports.

* * * * *

(c) * * *

(3) * * *

(v) *Hazard Index or HI.* The Hazard Index is an approach that determines the health concerns associated with mixtures of certain PFAS in finished drinking water. Low levels of multiple PFAS that individually would not likely result in adverse health effects may pose health concerns when combined in a mixture. The Hazard Index MCL represents the maximum level for mixtures of PFHxS, PFNA, HFPO–DA, and/or PFBS allowed in water delivered by a public water system. A Hazard Index greater than 1 requires a system to take action.

* * * * *

■ 11. Amend appendix A to subpart O, under the Contaminant heading ‘‘Synthetic organic contaminants including pesticides and herbicides:’’, by adding in alphabetical order entries for ‘‘Hazard Index PFAS (HFPO–DA, PFBS, PFHxS, and PFNA) (unitless)’’, ‘‘HFPO–DA (ng/l)’’, ‘‘PFHxS (ng/l)’’, ‘‘PFNA (ng/l)’’, ‘‘PFOA (ng/l)’’, and ‘‘PFOS (ng/l)’’ to read as follows:

Appendix A to Subpart O of Part 141—Regulated Contaminants

Contaminant (units)	Traditional MCL in mg/L	To convert for CCR, multiply by	MCL in CCR units	MCLG	Major sources in drinking water	Health effects language
*	*	*	*	*	*	*
Synthetic organic contaminants including pesticides and herbicides:	*	*	*	*	*	*
*	*	*	*	*	*	*
Hazard Index PFAS (HFPO–DA, PFBS, PFHxS, and PFNA) (unitless).	1 (unitless)	....	1	1	Discharge from manufacturing and industrial chemical facilities, use of certain consumer products, occupational exposures, and certain firefighting activities.	Per- and polyfluoroalkyl substances (PFAS) can persist in the human body and exposure may lead to increased risk of adverse health effects. Low levels of multiple PFAS that individually would not likely result in increased risk of adverse health effects may result in adverse health effects when combined in a mixture. Some people who consume drinking water containing mixtures of PFAS in excess of the Hazard Index (HI) MCL may have increased health risks such as liver, immune, and thyroid effects following exposure over many years and developmental and thyroid effects following repeated exposure during pregnancy and/or childhood.
*	*	*	*	*	*	*
HFPO–DA (ng/l).	0.00001	1,000,000	10	10	Discharge from manufacturing and industrial chemical facilities, use of certain consumer products, occupational exposures, and certain firefighting activities.	Some people who drink water containing HFPO–DA in excess of the MCL over many years may have increased health risks such as immune, liver, and kidney effects. There is also a potential concern for cancer associated with HFPO–DA exposure. In addition, there may be increased risks of developmental effects for people who drink water containing HFPO–DA in excess of the MCL following repeated exposure during pregnancy and/or childhood.

Contaminant (units)	Traditional MCL in mg/L	To convert for CCR, multiply by	MCL in CCR units	MCLG	Major sources in drinking water	Health effects language
*	*	*	*	*	*	*
PFHxS (ng/l)	0.00001	1,000,000	10	10	Discharge from manufacturing and industrial chemical facilities, use of certain consumer products, occupational exposures, and certain firefighting activities.	Some people who drink water containing PFHxS in excess of the MCL over many years may have increased health risks such as immune, thyroid, and liver effects. In addition, there may be increased risks of developmental effects for people who drink water containing PFHxS in excess of the MCL following repeated exposure during pregnancy and/or childhood.
PFNA (ng/l) ...	0.00001	1,000,000	10	10	Discharge from manufacturing and industrial chemical facilities, use of certain consumer products, occupational exposures, and certain firefighting activities.	Some people who drink water containing PFNA in excess of the MCL over many years may have increased health risks such as elevated cholesterol levels, immune effects, and liver effects. In addition, there may be increased risks of developmental effects for people who drink water containing PFNA in excess of the MCL following repeated exposure during pregnancy and/or childhood.
PFOA (ng/l) ...	0.0000040	1,000,000	4.0	0	Discharge from manufacturing and industrial chemical facilities, use of certain consumer products, occupational exposures, and certain firefighting activities.	Some people who drink water containing PFOA in excess of the MCL over many years may have increased health risks such as cardiovascular, immune, and liver effects, as well as increased incidence of certain types of cancers including kidney and testicular cancer. In addition, there may be increased risks of developmental and immune effects for people who drink water containing PFOA in excess of the MCL following repeated exposure during pregnancy and/or childhood.

Contaminant (units)	Traditional MCL in mg/L	To convert for CCR, multiply by	MCL in CCR units	MCLG	Major sources in drinking water	Health effects language
PFOS (ng/l) ...	0.0000040	1,000,000	4.0	0	Discharge from manufacturing and industrial chemical facilities, use of certain consumer products, occupational exposures, and certain firefighting activities.	Some people who drink water containing PFOS in excess of the MCL over many years may have increased health risks such as cardiovascular, immune, and liver effects, as well as increased incidence of certain types of cancers including liver cancer. In addition, there may be increased risks of developmental and immune effects for people who drink water containing PFOS in excess of the MCL following repeated exposure during pregnancy and/or childhood.
*	*	*	*	*	*	*

* * * * *

■ 12. Amend appendix A to subpart Q by:

■ a. Adding under the Contaminant heading "D. Synthetic Organic Chemicals (SOCs)" entries for "31", "32", "33", "34", "35", and "36" in numerical order;

■ b. Adding, immediately before footnote 1, footnote *; and

■ c. Adding footnote 23 at the end of the table.

The additions read as follows:

Appendix A to Subpart Q of Part 141—NPDWR Violations and Other Situations Requiring Public Notice[1]

Contaminant	MCL/MRDL/TT violations[2]		Monitoring & testing procedure violations	
	Tier of public notice required	Citation	Tier of public notice required	Citation
* * * *	*		*	*
D. Synthetic Organic Chemicals (SOCs)				
* * * *	*		*	*
31. Hazard Index PFAS	[23]*2	141.61(c)	3	141.905(c)
32. HFPO–DA	*2	141.61(c)	3	141.905(c)
33. PFHxS	*2	141.61(c)	3	141.905(c)
34. PFNA	*2	141.61(c)	3	141.905(c)
35. PFOA	*2	141.61(c)	3	141.905(c)
36. PFOS	*2	141.61(c)	3	141.905(c)
* * * *	*		*	*

Appendix A—Endnotes

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* Beginning April 26, 2029.

[1] Violations and other situations not listed in this table (*e.g.,* failure to prepare Consumer Confidence Reports), do not require notice, unless otherwise determined by the primacy agency. Primacy agencies may, at their option, also require a more stringent public notice tier (*e.g.,* Tier 1 instead of Tier 2 or Tier 2 instead of Tier 3) for specific violations and situations listed in this Appendix, as authorized under § 141.202(a) and § 141.203(a).

[2] MCL—Maximum contaminant level, MRDL—Maximum residual disinfectant level, TT—Treatment technique.

* * * * * * *

[23] Systems that violate the Hazard Index MCL and one or more individual MCLs based on the same contaminants may issue one notification to satisfy the public notification requirements for multiple violations pursuant to § 141.203.

■ 13. Amend appendix B to subpart Q by redesignating entries "55" through "89" as entries "61" through "95" and adding new entries "55" through "60" under the heading "E. Synthetic Organic Chemicals (SOCs)" to read as follows:

Appendix B to Subpart Q of Part 141—Standard Health Effects Language for Public Notification

Contaminant	MCLG[1] mg/L	MCL[2] mg/L	Standard health effects language for public notification
*	*	*	* * * *
E. Synthetic Organic Chemicals (SOCs)			
*	*	*	* * * *
55. Hazard Index PFAS (HFPO–DA, PFBS, PFHxS, and PFNA).	1 (unitless)	1 (unitless)	Per- and polyfluoroalkyl substances (PFAS) can persist in the human body and exposure may lead to increased risk of adverse health effects. Low levels of multiple PFAS that individually would not likely result in increased risk of adverse health effects may result in adverse health effects when combined in a mixture. Some people who consume drinking water containing mixtures of PFAS in excess of the Hazard Index (HI) MCL may have increased health risks such as liver, immune, and thyroid effects following exposure over many years and developmental and thyroid effects following repeated exposure during pregnancy and/or childhood.
56. HFPO–DA	0.00001	0.00001	Some people who drink water containing HFPO–DA in excess of the MCL over many years may have increased health risks such as immune, liver, and kidney effects. There is also a potential concern for cancer associated with HFPO–DA exposure. In addition, there may be increased risks of developmental effects for people who drink water containing HFPO–DA in excess of the MCL following repeated exposure during pregnancy and/or childhood.
57. PFHxS	0.00001	0.00001	Some people who drink water containing PFHxS in excess of the MCL over many years may have increased health risks such as immune, thyroid, and liver effects. In addition, there may be increased risks of developmental effects for people who drink water containing PFHxS in excess of the MCL following repeated exposure during pregnancy and/or childhood.
58. PFNA	0.00001	0.00001	Some people who drink water containing PFNA in excess of the MCL over many years may have increased health risks such as elevated cholesterol levels, immune effects, and liver effects. In addition, there may be increased risks of developmental effects for people who drink water containing PFNA in excess of the MCL following repeated exposure during pregnancy and/or childhood.
59. PFOA	Zero	0.0000040	Some people who drink water containing PFOA in excess of the MCL over many years may have increased health risks such as cardiovascular, immune, and liver effects, as well as increased incidence of certain types of cancers including kidney and testicular cancer. In addition, there may be increased risks of developmental and immune effects for people who drink water containing PFOA in excess of the MCL following repeated exposure during pregnancy and/or childhood.
60. PFOS	Zero	0.0000040	Some people who drink water containing PFOS in excess of the MCL over many years may have increased health risks such as cardiovascular, immune, and liver effects, as well as increased incidence of certain types of cancers including liver cancer. In addition, there may be increased risks of developmental and immune effects for people who drink water containing PFOS in excess of the MCL following repeated exposure during pregnancy and/or childhood.
*	*	*	* * * *

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[1] MCLG—Maximum contaminant level goal.

[2] MCL—Maximum contaminant level.

* * * * *

■ 14. Amend appendix C to subpart Q by adding entries for the acronyms "HI" and "PFAS" in alphabetical order to read as follows:

Appendix C to Subpart Q of Part 141—List of Acronyms Used in Public Notification Regulation

* * * * *

HI Hazard Index

* * * * *

PFAS Per- and Polyfluoroalkyl Substances

* * * * *

■ 15. Add subpart Z to read as follows:

Subpart Z—Control of Per- and Polyfluoroalkyl Substances (PFAS)

Sec.

Subpart Z—Control of Per- and Polyfluoroalkyl Substances (PFAS)

§ 141.900 General requirements.

(a) The requirements of this subpart constitute the national primary drinking water regulations for PFAS. Each community water system (CWS) and non-transient, non-community water system (NTNCWS) must meet the requirements of this subpart including the maximum contaminant levels for the PFAS identified in § 141.61(c).

(b) The deadlines for complying with the provisions of this subpart are as follows:

(1) Each system must meet the analytical requirements in § 141.901 by June 25, 2024.

(2) Each system must report the results of initial monitoring, as described in § 141.902(b)(1), to the State by April 26, 2027.

(3) Each system must meet the compliance monitoring requirements in § 141.902(b)(2) by April 26, 2027.

(4) Each system must meet the MCL compliance requirements in § 141.903 by April 26, 2029.

(5) Each system must meet the reporting and recordkeeping requirements in § 141.904 by April 26, 2027.

(6) Violations described in § 141.905 include monitoring and reporting violations and violations of MCLs. Monitoring and reporting violations may be assessed beginning on April 26, 2027. MCL violations may be assessed beginning on April 26, 2029.

§ 141.901 Analytical requirements.

(a) *General.* (1) Systems must use only the analytical methods specified in this section to demonstrate compliance with the requirements of this subpart.

(2) The following documents are incorporated by reference with the approval of the Director of the **Federal Register** in accordance with 5 U.S.C. 552(a) and 1 CFR part 51. This material is available for inspection at the EPA and at the National Archives and Records Administration (NARA). Contact the EPA's Drinking Water Docket at: 1301 Constitution Avenue NW., EPA West, Room 3334, Washington, DC 20460; phone: 202–566–2426. For information on the availability of this material at NARA, email: *fr.inspection@nara.gov*, or go to: *www.archives.gov/federal-register/cfr/ibr-locations.* The material may be obtained from the EPA at 1301 Constitution Avenue NW, the EPA West, Room 3334, Washington, DC 20460; phone: 202–566–2426; website: *https://www.epa.gov/pfas/epa-pfas-drinking-water-laboratory-methods.*

(i) EPA Method 533: Determination of Per- and Polyfluoroalkyl Substances in Drinking Water by Isotope Dilution Anion Exchange Solid Phase Extraction and Liquid Chromatography/Tandem Mass Spectrometry, 815–B–19–020, November 2019.

(ii) Method 537.1, Version 2.0: Determination of Selected Per- and Polyfluorinated Alkyl Substances in Drinking Water by Solid Phase Extraction and Liquid Chromatography/Tandem Mass Spectrometry (LC/MS/MS), EPA/600/R–20/006, March 2020.

(b) *PFAS*–(1) *Analytical methods.* Systems must measure regulated PFAS by the methods listed in the following table:

TABLE 1 TO PARAGRAPH (b)(1)—ANALYTICAL METHODS FOR PFAS CONTAMINANTS

Contaminant	Methodology	EPA method (incorporated by reference, see paragraph (a) of this section)
Perfluorobutane Sulfonate (PFBS)	SPE LC–MS/MS	533, 537.1, version 2.0.
Perfluorohexane Sulfonate (PFHxS)	SPE LC–MS/MS	533, 537.1, version 2.0.
Perfluorononanoate (PFNA)	SPE LC–MS/MS	533, 537.1, version 2.0.
Perfluorooctanesulfonic Acid (PFOS)	SPE LC–MS/MS	533, 537.1, version 2.0.
Perfluorooctanoic Acid (PFOA)	SPE LC–MS/MS	533, 537.1, version 2.0.
2,3,3,3-Tetrafluoro-2-(heptafluoropropoxy)propanoate (HFPO–DA or GenX Chemicals).	SPE LC–MS/MS	533, 537.1, version 2.0.

(2) *Laboratory certification.* Analyses under this section for regulated PFAS must only be conducted by laboratories that have been certified by EPA or the State. To receive certification to conduct analyses for the regulated PFAS, the laboratory must:

(i) Analyze Performance Evaluation (PE) samples that are acceptable to the State at least once during each consecutive 12-month period by each method for which the laboratory desires certification.

(ii) Beginning June 25, 2024, achieve quantitative results on the PE sample analyses that are within the following acceptance limits:

TABLE 2 TO PARAGRAPH (b)(2)(ii)—ACCEPTANCE LIMITS FOR PFAS PERFORMANCE EVALUATION SAMPLES

Contaminant	Acceptance limits (percent of true value)
Perfluorobutane Sulfonate (PFBS)	70–130
Perfluorohexane Sulfonate (PFHxS)	70–130
Perfluorononanoate (PFNA)	70–130
Perfluorooctanesulfonic Acid (PFOS)	70–130
Perfluorooctanoic Acid (PFOA)	70–130
2,3,3,3-Tetrafluoro-2-(heptafluoropropoxy)propanoate (HFPO–DA or GenX Chemicals)	70–130

(iii) For all samples analyzed for regulated PFAS in compliance with § 141.902, beginning June 25, 2024, report data for concentrations as low as the trigger levels as defined in § 141.902(a)(5).

§ 141.902 Monitoring requirements.

(a) *General requirements.* (1) Systems must take all samples during normal operating conditions at all entry points to the distribution system.

(2) If the system draws water from more than one source and the sources are combined before distribution, the system must sample at an entry point to the distribution system during periods of representative operating conditions.

(3) Systems must use only data collected under the provisions of this subpart to qualify for reduced monitoring.

(4) All new systems that begin operation after, or systems that use a new source of water after April 26, 2027, must demonstrate compliance with the MCLs within a period of time specified by the State. A system must also comply with initial sampling frequencies required by the State to ensure that the system can demonstrate compliance with the MCLs. Compliance monitoring frequencies must be conducted in accordance with the requirements in this section.

(5) For purposes of this section, the trigger levels are defined as shown in the following table.

TABLE 1 TO PARAGRAPH (a)(5)—TRIGGER LEVELS FOR PFAS CONTAMINANTS

Contaminant	Trigger level
Hazard Index PFAS (HFPO–DA, PFBS, PFHxS, PFNA).	0.5 (unitless).
HFPO–DA	5 nanograms per liter (ng/l).
PFHxS	5 ng/l.
PFNA	5 ng/l.
PFOA	2.0 ng/l.
PFOS	2.0 ng/l.

(6) Based on initial monitoring results, for each sampling point at which a regulated PFAS listed in § 141.61(c) is detected at a level greater than or equal to the trigger level, the system must monitor quarterly for all regulated PFAS beginning April 26, 2027, in accordance with paragraph (b)(2) of this section.

(7) For purposes of this section, each water system must ensure that all results provided by a laboratory are reported to the State and used for determining the required sampling frequencies. This includes values below the practical quantitation levels defined in § 141.903(f)(1)(iv); zero must not be used in place of reported values.

(b) *Monitoring requirements for PFAS*—(1) *Initial monitoring.* (i) Groundwater CWS and NTNCWS serving greater than 10,000 persons and all surface water CWS and NTNCWS must take four consecutive samples 2 to 4 months apart within a 12-month period (quarterly samples) for each regulated PFAS listed in § 141.61(c).

(ii) All groundwater CWS and NTNCWS serving 10,000 or fewer persons must take two samples for each regulated PFAS listed in § 141.61(c) five to seven months apart within a 12-month period.

(iii) All groundwater under the direct influence of surface water (GWUDI) CWS and NTNCWS must follow the surface water CWS and NTNCWS monitoring schedule in paragraph (b)(1)(i) of this section.

(iv) All systems that use both surface water and groundwater must apply the requirements in paragraphs (b)(1)(i) through (iii) of this section depending on the source(s) of water provided at a given entry point to the distribution system (EPTDS). If the EPTDS provides surface water, the requirements for a surface water CWS/NTNCWS apply. If the EPTDS provides groundwater, the requirements for a groundwater CWS/NTNCWS apply, based on system size. If an EPTDS provides a blend of surface water and groundwater, the requirements for a surface water system apply. For systems that change the source water type at an EPTDS during the initial monitoring period (*i.e.,* one part of the year it is surface water and the remaining part of the year it is groundwater), the sampling requirements for a surface water system apply.

(v) Systems must monitor at a frequency indicated in the following table, though a State may require more frequent monitoring on a system-specific basis:

TABLE 2 TO PARAGRAPH (b)(1)(v)—INITIAL MONITORING REQUIREMENTS

Type of system	Minimum monitoring frequency	Sample location
Groundwater CWS and NTNCWS serving greater than 10,000 persons, all surface water CWS and NTNCWS, and all GWUDI systems.	Four consecutive quarters of samples per entry point to the distribution system (EPTDS) within a 12-month period, unless the exception in paragraph (b)(1)(viii) of this section applies. Samples must be taken two to four months apart..	Sampling point for EPTDS.
Groundwater CWS and NTNCWS serving 10,000 or fewer persons.	Two consecutive samples per EPTDS within a 12-month period, unless the exception in paragraph (b)(1)(viii) of this section applies. Samples must be taken five to seven months apart..	Sampling point for EPTDS.

(vi) A State may accept data that has been previously acquired by a water system to count toward the initial monitoring requirements if the data meet the requirements of § 141.901(b)(1), samples were collected starting on or after January 1, 2019, and otherwise meet the timing requirements specified in table 2 to paragraph (b)(1)(v) of this section. For the purposes of satisfying initial monitoring requirements, acceptable data may be reported to a concentration no greater than the MCLs. However, a system is only eligible for triennial monitoring at the start of the compliance monitoring period if the system demonstrates that concentrations in all samples it uses to satisfy the initial monitoring requirements are below the trigger levels as defined in paragraph (a)(5) of this section.

(vii) If systems have multiple years of data, the most recent data must be used.

(viii) For systems using previously acquired data that have fewer than the number of samples required in a continuous 12-month period for initial monitoring as listed in table 2 to paragraph (b)(1)(v) of this section: All surface water systems, GWUDI systems, and groundwater systems serving greater than 10,000 persons must collect in a calendar year one sample in each quarter that was not represented, two to four months apart from the months with available data; All groundwater systems serving 10,000 or fewer persons must collect one sample in the month that is five to seven months apart from the month in which the previous sample was taken.

(ix) In determining the most recent data to report, a system must include all results provided by a laboratory whether above or below the practical quantitation levels. These results must be used for the purposes of determining the frequency with which a system must monitor at that sampling point at the start of the compliance monitoring period.

(x) States may delete results of obvious sampling errors. If the State deletes a result because of an obvious sampling error and the system fails to collect another sample this is a monitoring violation as described in § 141.905(c).

(xi) Initial monitoring requirements, including reporting results to the State, must be completed by April 26, 2027.

(2) *Compliance monitoring.* (i) Based on initial monitoring results, at the start of the monitoring period that begins on April 26, 2027, systems may reduce monitoring at each sampling point at which all reported sample concentrations were below all trigger levels defined in paragraph (a)(5) of this section, unless otherwise provided for by the State. At eligible sampling points,

each water system must analyze one sample for all regulated PFAS during each three-year monitoring period, at a time specified by the State, in the quarter in which the highest analytical result was detected during the most recent round of quarterly or semi-annual monitoring. If a sampling point is not eligible for triennial monitoring, then the water system must monitor quarterly at the start of the compliance monitoring period.

(ii) If, during the compliance monitoring period, a system is monitoring triennially and a PFAS listed in § 141.61(c) is detected at a level equal to or exceeding the trigger levels defined in paragraph (a)(5) of this section in any sample, then the system must monitor quarterly for all regulated PFAS beginning in the next quarter at the sampling point. The triggering sample must be used as the first quarter of monitoring for the running annual average calculation.

(iii) For all source water types, a State may determine that all regulated PFAS at a sampling point are reliably and consistently below the MCL after considering, at a minimum, four consecutive quarterly samples collected during the compliance monitoring period. A sampling point that a State has determined to be reliably and consistently below the MCL is required to collect annual samples for at least the first three years after that determination is made. Annual samples must be collected in the quarter in which detected concentrations were highest during the most recent year of quarterly monitoring. If, after three consecutive years, annual samples all contain results that are below the trigger levels defined in paragraph (a)(5) of this section, the State may allow a system to begin triennial monitoring at the sampling point. The water system must collect triennial samples in the quarter with the highest concentrations during the most recent round of quarterly sampling. If an annual sample meets or exceeds an MCL or the State determines that the result is not reliably and consistently below the MCL for all regulated PFAS, then the system must monitor quarterly for all regulated PFAS beginning in the next quarter at the sampling point.

(iv) The three different compliance monitoring sampling schedules that may be assigned and the criteria for each are summarized in the following table:

Table 3 to paragraph (b)(2)(iv)—Compliance Monitoring Schedules and Requirements

Sampling frequency	Eligibility requirements [1]	Sample timing requirements
Triennial	At an individual sampling point, either: (1) All initial monitoring results demonstrate concentrations of all regulated PFAS below trigger levels;. (2) The most recent three consecutive annual monitoring results all demonstrated concentrations of all regulated PFAS below trigger levels; or. (3) The previous triennial sample demonstrated all regulated PFAS concentrations below trigger levels.. **Note:** After beginning compliance monitoring, a system may not transition directly from quarterly monitoring to triennial monitoring..	Sample must be collected at a time within the three-year period designated by the State, in the quarter that yielded the highest analytical result during the most recent round of quarterly sampling (or the most recent semi-annual sampling, if no quarterly sampling has occurred).
Annual	A State makes a determination that all regulated PFAS concentrations at the sampling point are reliably and consistently below PFAS MCLs, after considering, at a minimum, 4 consecutive quarterly samples collected during the compliance monitoring period..	Sample must be collected at a time designated by the State, within the quarter that yielded the highest analytical result during the most recent round of quarterly sampling.
Quarterly	At an individual sampling point, either: (1) Any regulated PFAS concentration meets or exceeds a trigger level during initial monitoring;. (2) Sampling is occurring quarterly during compliance monitoring and a State has not made a determination that all levels of regulated PFAS at the sampling point are reliably and consistently below the regulated PFAS MCLs; or. (3) A sample collected by a system required to conduct triennial monitoring contains regulated PFAS concentrations that meet or exceed trigger levels. The first of these samples meeting or exceeding the trigger level is considered the first quarterly sample.. (4) A sample collected by a system required to conduct annual monitoring contains regulated PFAS concentrations that meet or exceed an MCL. The first of these samples meeting or exceeding the MCL is considered the first quarterly sample..	Samples must be collected in four consecutive quarters, on dates designated by the State.

[1] The monitoring frequency at a sampling point must be the same for all regulated PFAS and is determined based on the most frequent sampling required for any regulated PFAS detected at a level at or exceeding the trigger level.

(v) The State may require a confirmation sample for any sampling result. If a confirmation sample is required by the State, the system must average the result with the first sampling result and the average must be used for the determination of compliance with MCLs as specified by § 141.903. A State may delete results of obvious sampling errors from the MCL compliance calculations described in § 141.903. If the State deletes a result because of an obvious sampling error and the system fails to collect another sample this is a monitoring violation as described in § 141.905(c).

(vi) The State may increase the required monitoring frequency, where necessary, to detect variations within the system (*e.g.,* fluctuations in concentration due to seasonal use, changes in water source).

(vii) Each public water system must monitor at the time designated by the State within each monitoring period.

(viii) When a system reduces its sampling frequency to annual or triennial sampling, the next compliance sample must be collected in the monitoring period that begins the calendar year following State approval of a reduction in monitoring frequency.

§ 141.903 Compliance requirements.

(a) Compliance with MCLs for regulated PFAS in § 141.61(c) must be determined based on the analytical results obtained at each sampling point.

(b) For systems monitoring quarterly, compliance with the MCL is determined by the running annual average at each sampling point.

(c) If a system fails to collect the required number of samples specified in § 141.902, this is a monitoring violation as described in § 141.905(c), and compliance calculations must be based on the total number of samples collected.

(d) Systems monitoring triennially whose sample result equals or exceeds the trigger level of 2.0 ng/l for either PFOS or PFOA, 5 ng/l for HFPO–DA, PFHxS, or PFNA, or a Hazard Index of 0.5 for the Hazard Index PFAS, must begin quarterly sampling for all regulated PFAS in the next quarter at the sampling point. Systems monitoring annually whose sample result equals or exceeds the MCL of 4.0 ng/l for either PFOS or PFOA, 10 ng/l for HFPO–DA, PFHxS, or PFNA, or a Hazard Index of 1 for the Hazard Index PFAS, must begin quarterly sampling for all regulated PFAS in the next quarter at the sampling point.

(e) Except as provided in this paragraph (e), if a sample result exceeds an MCL, the system will not be considered in violation of the MCL until it has completed one year of quarterly sampling at the sampling point with the triggering sample used as the first quarter of monitoring for the running annual average calculation. However, whenever a sample result in any quarter (or quarterly average, if more than one compliance sample is available in a quarter because a confirmation sample was required by the State) causes the running annual average to exceed the MCL at a sampling point regardless of the subsequent quarterly monitoring results required to complete a full year of monitoring (*e.g.,* the results from a single sample are more than 4 times the MCL), the system is out of compliance with the MCL immediately.

(f) Systems must calculate compliance using the following method to determine MCL compliance at each sampling point:

(1) For each PFAS regulated by an individual MCL:

(i) For systems monitoring quarterly, divide the sum of the measured quarterly concentrations for each analyte by the number of quarters samples were collected for that analyte during the consecutive quarters included in the calculation. If more than one compliance sample for that analyte is available in a quarter because a confirmation sample was required by the State, systems must average all the results in a quarter then average the quarterly averages. Rounding does not occur until the end of the calculation. If the running annual average exceeds the MCL, the system is not in compliance with the MCL requirements.

(ii) For systems monitoring annually, if the concentration measured is equal to or exceeds an MCL for regulated PFAS, the system is required to initiate quarterly monitoring for all regulated PFAS beginning in the next quarter at the sampling point, with the triggering sample result used as the first quarter of monitoring for the running annual average calculation.

(iii) For systems monitoring triennially, if the concentration measured is equal to or exceeds the trigger level, the system is required to initiate quarterly monitoring for all regulated PFAS beginning in the next quarter at the sampling point, with the triggering sample result used as the first quarter of monitoring for the running annual average calculation.

(iv) For the purpose of calculating MCL compliance, if a sample result is less than the practical quantitation level (PQL) for a regulated PFAS, in accordance with the following table, zero is used for that analyte solely to calculate the running annual average.

TABLE 1 TO PARAGRAPH (f)(1)(iv)—PRACTICAL QUANTITATION LEVELS (PQLS) FOR PFAS CONTAMINANTS

Contaminant	PQL (in parts per trillion)
HFPO–DA	5.0
PFBS	3.0
PFHxS	3.0
PFNA	4.0
PFOA	4.0
PFOS	4.0

(2) For each PFAS regulated under the Hazard Index MCL:

(i) For systems monitoring quarterly, divide the observed sample analytical result for each analyte included in the Hazard Index by the corresponding HBWC listed in § 141.61(c) to obtain a hazard quotient for each analyte for each sampling event at each sampling point. Sum the resulting hazard quotients together to determine the Hazard Index for the quarter. If the State requires a confirmation sample for an analyte in the quarter, systems must average these results for each analyte in that quarter and then determine the hazard quotient(s) from those average values, then sum the hazard quotients. Once the Hazard Indices for the individual quarters are calculated, they are averaged to determine a running annual average. If the running annual average Hazard Index exceeds the MCL and two or more Hazard Index analytes had an observed sample analytical result at or above the PQL in any of the quarterly samples collected to determine the running annual average, the system is in violation of the Hazard Index MCL. No rounding occurs until after the running annual average Hazard Index is calculated.

(ii) If the Hazard Index calculated using the results of an annual sample equals or exceeds the Hazard Index MCL, the system must initiate quarterly sampling for all regulated PFAS beginning in the next quarter at the sampling point, with the triggering sample result used as the first quarter of monitoring.

(iii) If the Hazard Index calculated using the results of a triennial sample equals or exceeds the Hazard Index trigger level, the system must initiate quarterly sampling for all regulated PFAS beginning in the next quarter at the sampling point, with the triggering sample result used as the first quarter of monitoring.

(iv) If a sample result is less than the practical quantitation level for a regulated PFAS, in accordance with the table 1 to paragraph (f)(1)(iv) of this section, zero is used for that analyte solely to calculate the running annual average.

§ 141.904 Reporting and recordkeeping requirements.

Systems required to sample must report to the State according to the timeframes and provisions of § 141.31 and retain records according to the provisions in § 141.33.

(a) Systems must report the information from initial monitoring specified in the following table:

TABLE 1 TO PARAGRAPH (a)—DATA TO REPORT FROM INITIAL MONITORING

If you are a . . .	You must report . . .
System monitoring for regulated PFAS under the requirements of § 141.902(b)(1) on a quarterly basis.	1. All sample results, including the locations, number of samples taken at each location, dates, and concentrations reported. 2. Whether a trigger level, defined in § 141.902(a)(5), was met or exceeded in any samples.
System monitoring for regulated PFAS under the requirements of § 141.902(b)(1) less frequently than quarterly.	1. All sample results, including the locations, number of samples taken at each location, dates, and concentrations reported. 2. Whether a trigger level, defined in § 141.902(a)(5), was met or exceeded in any samples.

(b) Systems must report the information collected during the compliance monitoring period specified in the following table:

TABLE 2 TO PARAGRAPH (b)—DATA TO REPORT FROM COMPLIANCE MONITORING

If you are a . . .	You must report . . .
System monitoring for regulated PFAS under the requirements of § 141.902(b)(2) on a quarterly basis.	1. All sample results, including the locations, number of samples taken at each location, dates, and concentrations during the previous quarter. 2. The running annual average at each sampling point of all compliance samples. 3. Whether a trigger level, defined in § 141.902(a)(5), was met or exceeded in any samples. 4. Whether an MCL for a regulated PFAS in § 141.61(c) was met or exceeded in any samples. 5. Whether, based on § 141.903, an MCL was violated.
System monitoring for regulated PFAS under the requirements of § 141.902(b)(2) less frequently than quarterly.	1. All sample results, including the locations, number of samples taken at each location, dates, and concentrations during the previous monitoring period. 2. Whether a trigger level, defined in § 141.902(a)(5), was met or exceeded in any samples. 3. Whether an MCL for a regulated PFAS in § 141.61(c) was met or exceeded in any samples. 4. Whether, based on § 141.903, an MCL was violated (*e.g.,* the results from a single sample are more than 4 times the MCL).

§ 141.905 Violations.

(a) PFAS MCL violations, both for the individual PFOA, PFOS, HFPO–DA, PFHxS, and PFNA MCLs, as well as the Hazard Index MCL, as listed in § 141.61(c), are based on a running annual average, as outlined under § 141.903.

(b) Compliance with § 141.61(c) must be determined based on the analytical results obtained at each sampling point. If one sampling point is in violation of an MCL, the system is in violation of the MCL.

(c) Each failure to monitor in accordance with the requirements under § 141.902 is a monitoring violation.

(d) Failure to notify the State following a MCL violation and failure to submit monitoring data in accordance with the requirements of §§ 141.904 and 141.31 are reporting violations.

(e) Results for PFAS with individual MCLs as listed in § 141.61(c) are compared to their respective MCLs, and results for mixtures of two or more of the Hazard Index PFAS (HFPO–DA, PFBS, PFHxS, and PFNA) are compared to the Hazard Index MCL as listed in § 141.61(c). For determining compliance with the Hazard Index MCL, if only PFBS is reported at any concentration and no other regulated PFAS are in the mixture, it is not violation of the Hazard Index MCL. If only one of the other PFAS within the Hazard Index (HFPO–DA, PFHxS, and PFNA) is detected and the level of this PFAS exceeds its MCL as determined by § 141.903(f)(1)(i), only an individual MCL violation is assessed for the individual PFAS detected, and it is not a violation of the Hazard Index MCL. Exceedances of the Hazard Index caused by two or more of the Hazard Index PFAS (HFPO–DA, PFBS, PFHxS, and PFNA) and exceedances of one or more individual MCLs can result in multiple MCL exceedances. However, in this instance, for purposes of public notification under appendix A to subpart Q of this part, a PWS must only report the Hazard Index MCL exceedance.

PART 142—NATIONAL PRIMARY DRINKING WATER REGULATIONS IMPLEMENTATION

■ 16. The authority citation for part 142 continues to read as follows:

Authority: 42 U.S.C. 300f, 300g–1, 300g–2, 300g–3, 300g–4, 300g–5, 300g–6, 300j–4, 300j–9, and 300j–11.

■ 17. Amend § 142.16 by adding paragraph (r) to read as follows:

§ 142.16 Special primacy requirements.

* * * * *

(r) *Requirements for States to adopt 40 CFR part 141, subpart Z, PFAS.* In addition to the general primacy requirements elsewhere in this part, including the requirements that State regulations be at least as stringent as Federal requirements, an application for approval of a State program revision that adopts 40 CFR part 141, subpart Z, must contain the following, in lieu of meeting the requirements of paragraph (e) of this section:

(1) The State's procedures for reviewing the water system's use of pre-existing data to meet the initial

monitoring requirements specified in § 141.902, including the criteria that will be used to determine if the data are acceptable. This paragraph (r)(1) is no longer applicable after the initial monitoring period ends on April 26, 2027.

(2) The State's procedures for ensuring all systems complete the initial monitoring period requirements that will result in a high degree of monitoring compliance by the regulatory deadlines. This paragraph (r)(2) is no longer applicable after the initial monitoring period ends on April 26, 2027.

(3) After the initial monitoring period, States establish the initial monitoring requirements for new public water systems and existing public water systems that plan to use a new source. States must explain their initial monitoring schedules and how these monitoring schedules ensure that new public water systems and existing public water systems that plan to use new sources comply with MCLs and monitoring requirements. States must also specify the time frame in which a new system or existing system that plans to use a new source must demonstrate compliance with the MCLs.

■ 18. Amend § 142.62 by revising and republishing paragraph (a) to read as follows:

§ 142.62 Variances and exemptions from the maximum contaminant levels for organic and inorganic chemicals.

(a) The Administrator, pursuant to section 1415(a)(1)(A) of the Act, hereby identifies the technologies listed in tables 1 and 2 to this paragraph (a) as the best available technology, treatment techniques, or other means available for achieving compliance with the maximum contaminant levels for the organic chemicals, including per- and polyfluoroalkyl substances (PFAS), listed in § 141.61(a) and (c) of this chapter, for the purposes of issuing variances and exemptions. A list of small system compliance technologies for the regulated PFAS for the purposes of providing variances and exemptions is provided in table 3 to this paragraph (a); for the purpose of this paragraph (a), small system is defined as a system serving 10,000 persons or fewer.

TABLE 1 TO PARAGRAPH (a)—BATS FOR PFAS LISTED IN § 141.61(c)

Contaminant	BAT
Hazard Index PFAS (HFPO–DA, PFBS, PFHxS, and PFNA)	Anion exchange, GAC, reverse osmosis, nanofiltration.
HFPO–DA	Anion exchange, GAC, reverse osmosis, nanofiltration.
PFHxS	Anion exchange, GAC, reverse osmosis, nanofiltration.
PFNA	Anion exchange, GAC, reverse osmosis, nanofiltration.
PFOA	Anion exchange, GAC, reverse osmosis, nanofiltration.
PFOS	Anion exchange, GAC, reverse osmosis, nanofiltration.

TABLE 2 TO PARAGRAPH (a)—BATS FOR OTHER SYNTHETIC ORGANIC CONTAMINANTS LISTED IN § 141.61(c) AND VOLATILE ORGANIC CHEMICALS LISTED IN § 141.61(a)

Contaminant	Best available technologies		
	PTA[1]	GAC[2]	OX[3]
(1) Benzene	X	X	
(2) Carbon tetrachloride	X	X	
(3) 1,2-Dichloroethane	X	X	
(4) Trichloroethylene	X	X	
(5) para-Dichlorobenzene	X	X	
(6) 1,1-Dichloroethylene	X	X	
(7) 1,1,1-Trichloroethane	X	X	
(8) Vinyl chloride	X		
(9) cis-1,2-Dichloroethylene	X	X	
(10) 1,2-Dichloropropane	X	X	
(11) Ethylbenzene	X	X	
(12) Monochlorobenzene	X	X	
(13) o-Dichlorobenzene	X	X	
(14) Styrene	X	X	
(15) Tetrachloroethylene	X	X	
(16) Toluene	X	X	
(17) trans-1,2-Dichloroethylene	X	X	
(18) Xylense (total)	X	X	
(19) Alachlor	....	X	
(20) Aldicarb	....	X	
(21) Aldicarb sulfoxide	....	X	
(22) Aldicarb sulfone	....	X	
(23) Atrazine	....	X	
(24) Carbofuran	....	X	
(25) Chlordane	....	X	
(26) Dibromochloropropane	X	X	
(27) 2,4-D	....	X	
(28) Ethylene dibromide	X	X	
(29) Heptachlor	....	X	
(30) Heptachlor epoxide	....	X	
(31) Lindane	....	X	
(32) Methoxychlor	....	X	
(33) PCBs	....	X	
(34) Pentachlorophenol	....	X	

TABLE 2 TO PARAGRAPH (a)—BATS FOR OTHER SYNTHETIC ORGANIC CONTAMINANTS LISTED IN § 141.61(c) AND VOLATILE ORGANIC CHEMICALS LISTED IN § 141.61(a)—Continued

Contaminant	Best available technologies		
	PTA [1]	GAC [2]	OX [3]
(35) Toxaphene		X	
(36) 2,4,5-TP		X	
(37) Benzo[a]pyrene		X	
(38) Dalapon		X	
(39) Dichloromethane	X		
(40) Di(2-ethylhexyl)adipate	X	X	
(41) Di(2-ethylhexyl)phthalate		X	
(42) Dinoseb		X	
(43) Diquat		X	
(44) Endothall		X	
(45) Endrin		X	
(46) Glyphosate			X
(47) Hexachlorobenzene		X	
(48) Hexachlorocyclopentadiene	X	X	
(49) Oxamyl (Vydate)		X	
(50) Picloram		X	
(51) Simazine		X	
(52) 1,2,4-Trichlorobenzene	X	X	
(53) 1,1,2-Trichloroethane	X	X	
(54) 2,3,7,8-TCDD (Dioxin)		X	

[1] Packed Tower Aeration.

[2] Granular Activated Carbon.

[3] Oxidation (Chlorination or Ozonation).

TABLE 3 TO PARAGRAPH (a)—LIST OF SMALL SYSTEM COMPLIANCE TECHNOLOGIES (SSCTS) [1] FOR PFAS LISTED IN § 141.61(c)

Small system compliance technologies	Affordable for listed small system categories [2]
Anion Exchange	All size categories.
GAC	All size categories.
Reverse Osmosis,[3] Nanofiltration [3].	3,301–10,000.

[1] Section 1412(b)(4)(E)(ii) of SDWA specifies that SSCTs must be affordable and technically feasible for small systems.

[2] The Act (ibid.) specifies three categories of small systems: (i) those serving 25 or more, but fewer than 501, (ii) those serving more than 500, but fewer than 3,301, and (iii) those serving more than 3,300, but fewer than 10,001.

[3] Technologies reject a large volume of water and may not be appropriate for areas where water quantity may be an issue.

* * * * *

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TECHNICAL FACT SHEET

Draft National Recommended Human Health Ambient Water Quality Criteria for PFOA, PFOS, and PFBS

December 2024

As part of the Environmental Protection Agency's efforts to safeguard human health from exposure to per- and polyfluoroalkyl substances (PFAS), the agency has published draft national recommended human health criteria for three PFAS – perfluorooctanoic acid (PFOA), perfluorooctane sulfonic acid (PFOS), and perfluorobutane sulfonic acid (PFBS) – for a 60-day public comment period. These draft Clean Water Act (CWA) recommended criteria reflect the latest scientific knowledge regarding the human health effects, exposure information, and bioaccumulation potential. Once final, the human health criteria will provide national recommendations to states and Tribes authorized to establish their own water quality standards under the Clean Water Act. National recommended human health ambient water quality criteria are not regulations, nor do they impose legally binding requirements.

Background

To help states and Tribes protect people from harmful health effects from exposure to pollutants in surface waters, the EPA periodically publishes national human health criteria recommendations under CWA Section 304(a). The national recommended human health criteria provide information for states and authorized Tribes to consider when developing their own water quality standards.

On October 18, 2021, EPA Administrator Regan announced the agency's *PFAS Strategic Roadmap*, laying out a whole-of-agency approach to addressing PFAS. This included the EPA's efforts to develop PFAS recommended human health criteria (HHC), starting with PFOA and PFOS. The EPA has released draft recommended human health criteria for PFOA, PFOS, and PFBS and will accept public comments on these criteria documents for 60 days upon announcement in the Federal Register. Following the comment period, the EPA will review the public comments and revise the criteria accordingly. The EPA will then release final criteria recommendations and responses to public comments.

What are national recommended human health water quality criteria?

People can be exposed to pollutants, such as PFAS, when drinking water and eating fish and shellfish from polluted water bodies. Under CWA Section 304(a), the EPA develops, and from time to time, updates, recommended water quality criteria for the protection of human health. States and authorized Tribes can consider the EPA's recommended criteria when setting water quality standards for their lakes, rivers, and other inland and nearshore water bodies to protect public health.

The EPA's CWA Section 304(a) HHC documents provide scientific information on the human health effects of the pollutants as well as the national recommended levels of pollutants in water (criteria), which, if not exceeded, are expected to protect against adverse effects to human health. For each contaminant, the agency has derived two HHC values: (1) the "water + organism" HHC and (2) the "organism only" HHC. The EPA develops national recommended human health criteria using a mathematical equation that includes a final toxicity factor, exposure factors (for body weight, fish consumption, and drinking water intake), bioaccumulation factors, and a relative source contribution which accounts for exposure from other potential sources (i.e., air, soils, marine fish consumption). The equations for deriving water + organism, or organism only criteria, as well as equations for cancer and non-cancer-based HHC, can be found in the EPA's 2000 Methodology (EPA, 2000) document and in each HHC document.

What are the draft National Recommended Ambient Water Quality Criteria for the Protection of Human Health for PFOA, PFOS, and PFBS?

The draft criteria (Table 1) summarize the ambient concentrations for PFOA, PFOS, or PFBS in surface water, which, if not exceeded will protect the general population from adverse health effects due to ingesting water, fish, and shellfish from inland and nearshore water bodies. The EPA provides recommendations for "water + organism" and "organism only" criteria for states and authorized Tribes to consider when adopting human health criteria into their water quality standards. Under the CWA, states and authorized Tribes designate the uses of their water bodies and adopt criteria into their water quality standards to support those uses.

For PFOA and PFOS, the EPA developed draft recommended HHC based on the final non-cancer toxicity values (RfD) and cancer slope factors (CSF), as both chemicals are *Likely to be Carcinogenic to Humans*, based on the final human health toxicity assessments (EPA, 2024a, b). For PFOA, the EPA is recommending HHC based on the values derived using cancer inputs because they are lower than the noncancer-based HHC and thus, protect against both adverse noncancer and cancer health effects. For PFOS, the EPA is recommending the noncancer HHC because they are lower and thus, protective of both adverse noncancer and cancer health effects. For PFBS, the draft recommended human health criteria are based only on non-cancer health effects because the PFBS toxicity assessment determined that there is *Inadequate Information to Assess Carcinogenic Potential* for PFBS and no CSF was developed.

Under the EPA's recently finalized Method 1633 used for analysis of these three PFAS in aqueous samples, the limit of quantification (LOQ) representing the observed LOQs in the multi-laboratory validation study, range from 1 to 4 ng/L. The pooled Method Detection Limits (MDLs) are 0.54 ng/L (PFOA), 0.63 ng/L (PFOS), and 0.37 ng/L (PFBS). The pooled MDL values are derived from the multi-laboratory validation study using MDL data from eight laboratories and represent the sensitivity that should be achievable in a well-prepared laboratory but may not represent the actual MDL used for data reporting or data quality assessments (EPA, 2024c). The MDLs and ranges presented here are provided for comparison of analytical concentrations and draft recommended HHC.

Table 1. Draft Human Health Criteria (HHC) for Three PFAS.

PFAS	Water + Organism HHC (ng/L; ppt)[1]	Organism Only HHC (ng/L; ppt)[1]
PFOA	0.0009	0.0036
PFOS	0.06	0.07
PFBS	400	500

[1] Values are provided in ng/L units to aid in comparison to method detection limit (MDL).

Consideration of PFAS mixtures for HHC

The available scientific information shows that mixtures of PFAS can result in dose additive health effects, meaning that the combined effect of the component chemicals in a mixture is equal to the sum of the individual doses or concentrations, scaled for potency. Therefore, specific approaches based on dose-additivity are available and could be used to the develop a PFAS mixture HHC. In these draft recommended criteria, the EPA provides an illustrative example for states or Tribes interested in developing a water quality standard for a mixture of two or more PFAS for which human health criteria have been developed using the hazard index (HI) approach. The HI approach is further described in the final *Framework for Estimating Noncancer Health Risks Associated with Mixtures of Per- and Polyfluoroalkyl Substances (PFAS)*.

Where can I find more information?

View the draft human health criteria documents for PFOA, PFOS, and PFBS, and other related information on the EPA's website at: https://www.epa.gov/wqc/human-health-water-quality-criteria-pfas
For more information on water quality criteria, visit: https://www.epa.gov/wqc
For more information on PFAS, visit: https://www.epa.gov/pfas

References

EPA. 2000. *Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health (2000).* EPA 822-B-00-004. Available on the internet at: https://www.epa.gov/sites/default/files/2018-10/documents/methodology-wqc-protection-hh-2000.pdf.

EPA. 2024a. *Final Human Health Toxicity Assessment for Perfluorooctanoic Acid (PFOA) and Related Salts*. EPA 815R24006. Office of Water, Washington, DC. Available on the internet at: https://www.epa.gov/system/files/documents/2024-05/final-human-health-toxicity-assessment-pfoa.pdf.

EPA. 2024b. *Final Human Health Toxicity Assessment for Perfluorooctane Sulfonic Acid (PFOS) and Related Salts*. EPA 815R24007. Office of Water, Washington, DC. Available on the internet at: https://www.epa.gov/system/files/documents/2024-05/final-human-health-toxicity-assessment-pfos.pdf.

EPA. 2024c. Method 1633. Analysis of Per-and Polyfluoroalkyl Substances (PFAS) in Aqueous, Solid, Biosolids, and Tissue Samples by LC-MS/MS. EPA 821-R-24-001. Office of Water, Washington, DC. https://www.epa.gov/system/files/documents/2024-12/method-1633a-december-5-2024-508-compliant.pdf.