American Water Works Association
Dedicated to the World's Most Important Resource™

Government Affairs Office
1300 Eye Street NW
Suite 701W
Washington, DC 20005-3314
T 202.628.8303
F 202.628.2846

May 30, 2023

Michael Regan
Administrator
Environmental Protection Agency
1200 Pennsylvania Avenue, N.W.
Mail Code: 1309
Washington, DC 20004

SUBMITTED ELECTRONICALLY

RE: Per- and polyfluoroalkyl substances (PFAS): Perfluorooctanoic acid (PFOA) and Perfluorooctanesulfonic acid (PFOS) National Primary Drinking Water Regulation Rulemaking, Docket No. EPA–HQ–OW-2022-0114

Dear Administrator Regan,

The American Water Works Association appreciates the U.S. Environmental Protection Agency's (EPA) efforts to propose national primary drinking water regulations for per- and polyfluoroalkyl substances (PFAS), including perfluorooctanoic acid (PFOA) and perfluorooctanesulfonic acid (PFOS). The proposal includes several major actions for PFAS in drinking water, including:

- Proposal for drinking water standards for PFOA and PFOS, individually,
- Preliminary determinations for perfluorohexanesulfonic acid (PFHxS), perfluorononanoic acid (PFNA), hexafluoropropylene oxide dimer acid (HFPO-DA), perfluorobutanesulfonic acid (PFBS), and the mixture of these four PFAS, and
- Proposal for drinking water standard for PFHxS, PFNA, HFPO-DA, and PFBS as a mixture using a hazard index.

AWWA supports the development of primary drinking water standards for PFOA and PFOS and supports the agency's interest in proposing regulatory determinations for additional PFAS. AWWA recommended the development of standards for PFOA and PFOS in comments to the EPA in 2021 and provided a shortlist of PFAS compounds for the agency's consideration for additional action as appropriate. In these comments, AWWA provided additional recommendations relating to the use of occurrence data, an approach to monitoring requirements, and available cost data for drinking water treatment facilities.

AWWA believes that EPA has put forward a rule framework that begins to address a number of stakeholder concerns. The proposal serves as a good starting point for finalizing a rule that will address PFAS compounds in drinking water. Attached are detailed comments on the proposed rule. In addition to providing feedback for improving the analyses to support the proposal, AWWA makes several key recommendations, which are further detailed in the comment letter.

These recommendations include:

1. The agency should consider withdrawing and re-proposing drinking water standards for PFOA and PFOS given the recurring issues with the underlying analyses. If the agency should finalize drinking water standards for PFOA and PFOS based on the current proposal, drinking water standards of 10 ppt, each, are most appropriate.
2. The agency's preliminary determinations for PFNA, HFPO-DA, and PFBS and the mixture of PFHxS, PFNA, HFPO-DA, and PFBS are not sufficiently supported by the available data. Moreover the available data currently suggests a negative determination for these PFAS is appropriate. The agency should re-issue these preliminary determinations following the availability of national monitoring data currently being collected, as part of the Fifth Unregulated Contaminant Monitoring Rule (UCMR 5).
3. The agency misuses the hazard index as a maximum contaminant level given it is not supported by federal guidance for assessing risk from mixtures and other issues.
4. The agency's proposed regulation concurrent with preliminary regulatory determinations is not within the scope of authorities granted by the Safe Drinking Water Act (SDWA), does not fulfill the obligations under the Administrative Procedures Act, and is inappropriate. Proposed regulation of additional PFAS should not occur until a determination to regulate is issued.

We hope that these comments will help EPA finalize the rule by effectively leveraging science and the authorities of the Safe Drinking Water Act. If you have any questions regarding this correspondence, please contact me or Chris Moody at 202.326.6127 or cmoody@awwa.org.

Best Regards,
ON BEHALF OF THE AMERICAN WATER WORKS ASSOCIATION

G. Tracy Mehan, III
Executive Director for Government Affairs

Attachment (1)

cc: Ryan Albert, EPA / OW
Radhika Fox, EPA / OW
Eric Burneson, EPA / OW
Jennifer McLain, EPA / OW

Who is AWWA

The American Water Works Association is an international, nonprofit, scientific and educational society dedicated to providing total water solutions assuring the effective management of water. Founded in 1881, the Association is the largest organization of water supply professionals in the world. Our membership includes more than 4,500 utilities that supply roughly 80 percent of the nation's drinking water and treat almost half of the nation's wastewater. Our 50,000-plus total membership represents the full spectrum of the water community: public water and wastewater systems, environmental advocates, scientists, academicians, and others who hold a genuine interest in water, our most important resource. AWWA unites the diverse water community to advance public health, safety, the economy, and the environment.

AWWA Comments on the Proposed "PFAS National Primary Drinking Water Regulation Rulemaking"

Docket ID No: EPA-HQ-OW-2022-0114

Prepared by the:
American Water Works Association

Table of Contents

List of Appendices

List of Tables

List of Figures

List of Acronyms

APA	Administrative Procedures Act
ATSDR	Agency for Toxic Substances and Disease Registry
AWWA	American Water Works Association
BIL	Bipartisan Infrastructure Law
CCT	Corrosion Control Treatment
CVD	Cardiovascular Disease
CWA	Clean Water Act
DBP	Disinfection Byproduct
DWSRF	Drinking Water State Revolving Fund
ELG	Effluent Limitation Guideline and Standard
EPA	Environmental Protection Agency
GAC	Granular Activated Carbon
HAL	Health Advisory Level
HBWC	Health Based Water Concentration
HFPO-DA	Hexafluoropropylene Oxide Dimer Acid
HRRCA	Health Risk Reduction and Cost Analysis
IRIS	Integrated Risk Information System
IX	Ion Exchange
LCRR	Lead and Copper Rule Revisions
MCL	Maximum Contaminant Level
MCLG	Maximum Contaminant Level Goal
NDAA 2020	National Defense Authorization Act for Fiscal year 2020
NHANES	National Health and Nutrition Examination Survey
NPDWR	National Primary Drinking Water Regulation
PFAS	Per- and Polyfluoroalkyl Substance
PFBS	Perfluorobutanesulfonic Acid
PFHxS	Perfluorohexanesulfonic Acid
PFNA	Perfluorononanoic Acid
PFOA	Perfluorooctanoic Acid
PFOS	Perfluorooctanesulfonic Acid
PQL	Practical Quantification Level
RAA	Running Annual Average
RCRA	Resource Conservation and Recovery Act
RO	Reverse Osmosis
SAB	Science Advisory Board
SDWA	Safe Drinking Water Act
SMF	Standard Monitoring Framework
TOC	Total Organic Carbon
TRI	Toxics Release Inventory
TSCA	Toxic Substances Control Act
UCMR 3	Third Unregulated Contaminant Monitoring Rule
UCMR 5	Fifth Unregulated Contaminant Monitoring Rule
WBS	Work Breakdown Structure
WIFIA	Water Infrastructure Finance and Innovation Act

AWWA Comments on the

Proposed "PFAS National Primary Drinking Water Regulation Rulemaking"

The American Water Works Association (AWWA) appreciates the opportunity to comment on the Environmental Protection Agency's (EPA or agency) "Proposed Per- and polyfluoroalkyl substances (PFAS) National Primary Drinking Water Regulation Rulemaking" (the proposal). AWWA has prepared the following comments to assist EPA in moving forward with a final rule to protect public health that is scientifically grounded, legally defensible, and crafted in a manner that embraces the objectives of the Safe Drinking Water Act (SDWA).

1. Overarching Comments

AWWA appreciates EPA's interest in addressing PFAS in drinking water to protect public health and maintain public trust in the nation's drinking water supply. AWWA has been engaged on PFAS issues since the early 2000s and has leveraged technical expertise from our members, which include more than 50,000 professional members and 4,500 utilities, to support the agency's broad efforts to address PFAS contamination. When the following guiding principles are followed, AWWA supports PFAS regulations:

1. Commitment to public health protection,
2. Fidelity to scientific process,
3. Setting regulatory requirements that are feasible to implement,
4. Ensuring affordability of safe drinking water, and
5. Effectively leveraging source water protection efforts.

In establishing drinking water regulations, embracing these guiding principles will ensure that communities and the public are effectively protected through a transparent rulemaking process and with a rule that prioritizes opportunities to reduce public health risks. The following comments and recommendations reflect these guiding principles.

Reinforcing the Polluter Pays Principle

EPA first published plans for a broad regulatory agenda to address PFAS as part of the EPA's PFAS Action Plan in 2019 (EPA, 2019a). In 2021, the agency published the more detailed PFAS Strategic Roadmap for regulatory actions (EPA, 2021a). The EPA's PFAS Action Plan and the PFAS Strategic Roadmap highlighted a variety of regulatory actions that the agency is pursuing, including setting effluent limitation guidelines and standards (ELGs) for industrial dischargers under the Clean Water Act (CWA). The implementation of these regulatory actions is critical in protecting the environment and the protection of drinking water sources. The actions, when completed, will reinforce the polluter pays principle for PFAS and help maintain the responsibility for the mitigation of PFAS contamination on the polluters instead of communities. While NPDWRs will require community investment to address PFAS, actions under the CWA, Resource Conservation and Recovery Act (RCRA), and the Toxic Substances Control Act (TSCA) will require manufacturers and users of PFAS to carry this burden.

To date, however, the agency's actions on polluters have consistently lagged behind drinking water action. EPA originally identified PFAS as a potential priority for drinking water as part of the Contaminant Candidate List 3 in 2009 (EPA, 2009). In 2012 EPA advanced the Third Unregulated Contaminant Monitoring Rule (UCMR 3) that required water systems to monitor for six PFAS in finished drinking water

(EPA, 2012). With the proposal, EPA is proposing to set standards for PFAS in drinking water. At the same time, EPA has yet to advance regulations that require manufacturers and users to: (i) report about uses and releases of PFAS, (ii) control the release of PFAS to the environment, (iii) manage PFAS-containing wastes appropriately, and (iv) limit the use and manufacturing of PFAS (EPA, 2022a; EPA, 2022b; EPA, 2022c).

What is further concerning, is the lack of urgency in advancing these actions by the agency. The TSCA data reporting rule, which will require manufacturers and users to report on the production, use, and release of PFAS, was prompted by Congress as part of the National Defense Authorization Act for Fiscal Year 2020 (NDAA 2020) in December 2019 and proposed in June 2021 (Congress, 2019; EPA, 2022a). The rule has yet to be finalized, despite a statutory deadline of January 2023. Additionally, EPA initiated an effort under the CWA to consider ELGs for PFAS as part of the Preliminary Effluent Guidelines Program Plan 14 (EPA, 2019b). With this plan, EPA committed to performing a study of PFAS in industrial effluents for several industries. Program Plan 15 moved forward with a commitment to initiate two rulemakings for both manufacturers and metal finishers and to initiate additional studies on landfills and textile mills (EPA, 2021b). Neither of these rulemakings have been proposed. These actions, if advanced with the same sense of urgency as drinking water actions, would have provided invaluable information and protection for PFAS releases to the environment and the drinking water sources.

The responsibility of the Administrator is to ensure that regulatory actions are implemented in a cohesive manner for the effective protection of the environment and the public. It is imperative that the Administrator begin to advance these actions more meaningfully to minimize the role that communities play in addressing PFAS contamination that they were not responsible for causing. Advancing these actions in a more meaningful, cohesive manner has the potential to curb costly burdens on water system rate payers.

Ensuring Community Resources are Invested in High Priorities

In crafting NPDWRs, it is imperative that the abovementioned guiding principles be followed by the agency to ensure that community resources are invested in high priority risks. In crafting SDWA, Congress recognized the importance of addressing contaminants of greatest concern as part of the development of NPDWRs. EPA must consider the impact this final rule will have on communities as they are managing multiple priorities for community investments to protect public health. Important examples include the replacement of lead service lines, enhancing cybersecurity protections, continuing to improve risk reductions related to disinfection byproducts (DBPs), and the continuous efforts to replace and maintain aging infrastructure to avoid the risk of water main breaks and other threats to public health. The benefits and costs of new standards must be carefully, and accurately, weighed to ensure the investments needed to meet new regulatory requirements do not inappropriately lead to re-allocating available funds away from public health concerns of higher priority, causing unintended consequences.

A recent analysis by Black & Veatch estimated that the costs of the proposed standards could exceed $2.5 to $3.2 billion annually (Black & Veatch, 2023 – See Appendix B). The Administrator will need to determine if these costs are justified by the benefits, estimated to be $0.8 to $1.2 billion annually, and whether it is a meaningful opportunity to protect public health when this investment will divert water systems investments from other needs to assuring compliance with any final PFAS rule requirements.

Summary of Key Recommendations

AWWA reviewed all aspects of the proposal and the supporting documentation, including the agency's occurrence analysis, cost analysis, benefits analysis, and household affordability analysis. The proposal includes several major actions for PFAS in drinking water, including:

1. Proposal for drinking water standards for both PFOA and PFOS;
2. Preliminary determinations for perfluorohexanesulfonic acid (PFHxS), perfluorononanoic acid (PFNA), hexafluoropropylene oxide dimer acid (HFPO-DA), perfluorobutanesulfonic acid (PFBS), and the mixture of these four PFAS, and
3. Proposal for drinking water standard for PFHxS, PFNA, HFPO-DA, and PFBS as a mixture using a hazard index.

Based on the supporting documentation, and in consultation with drinking water technical experts, AWWA recommends that the agency consider withdrawing and re-proposing the drinking water standards for PFOA and PFOS given that the underlying analyses lack transparency, are not consistent with use of best available science, and are not clear. However, if EPA issues a final rule setting standards PFOA and PFOS, the agency should set drinking water standards of 10 ppt PFOA and 10 ppt PFOS on the basis that these would be most defensible with the agency's current analysis.

The agency's preliminary determinations for PFNA, HFPO-DA, and PFBS are not sufficiently supported by the supporting documentation. The information included in the proposed rule docket does not suggest that there is a substantial likelihood of PFNA, HFPO-DA, and PFBS occurrence in drinking water with a frequency and at levels of public health concern. Instead, the available evidence indicates that a negative determination is appropriate. Similarly, the information on co-occurrence and overall occurrence of PFHxS, PFNA, HFPO-DA, and PFBS as a mixture also indicates that the agency only has supporting information for a negative determination. Preliminary determinations for these compounds and their mixture should be re-issued following completion of the Fifth Unregulated Contaminant Monitoring Rule (UCMR 5).

Additionally, the agency's proposed drinking water standard for the mixture of PFHxS, PFNA, HFPO-DA, and PFBS poses several issues that must be addressed prior to further action. As discussed above, the underlying data to support a regulatory determination is insufficient and likely suggests that regulation does not represent meaningful opportunity to protect public health. The agency's approach to using a general hazard index with multiple health outcomes lacks support from risk assessment guidance and professionals (ATSDR, 2018; ATSDR, 2022; EPA, 1986; EPA, 2000; SAB, 2022). Finally, the proposed regulation concurrent with preliminary determinations is not within the scope of the EPA's authority under SDWA. While EPA is authorized to issue a proposal concurrent with a determination to regulate, this is a distinctly different action from a preliminary determination.

Additional recommendations are detailed in this letter, including:

1. The proposal's underlying cost analysis is inaccurate and substantially underestimates costs to compliance costs and the financial impacts on consumers. The agency should work with AWWA and other stakeholders to develop a cost-estimating approach that is more reflective of best-engineering practices and should refine the cost estimate for the proposal. Any final rule should

re-evaluate the determinations that benefits justify the costs and the rule is affordable for small systems, considering the updated cost analysis to support any rule.

2. The standard implementation timeline is not appropriate for the current economic conditions. The process to monitor, plan, design, pilot, permit, and construct facilities will take much longer than the standard 3-year timeline given workforce and supply chain challenges. While states may be able to grant 2-year extensions, on a case-by-case basis the burden for state primacy agencies to process these requests will be significant. To avoid a final rule that is infeasible for water systems to comply with, the Administrator should exercise its authority under the SDWA to extend the effective date of compliance by two years for water systems installing capital improvements[1].

3. EPA's statistical approach to estimating occurrence of PFAS in drinking water is overly complicated and EPA would be ill-advised to move forward with a final rule without considering incoming data from more than 3,500 water systems currently collecting samples under the Fifth UCMR 5 (EPA, 2021c).

4. The results of the health risk reduction and cost analysis (HRRCA) are mixed and demonstrate the uncertainty of actual outcomes on communities. Under one analysis (3% discount rate), net benefits are expected for each rule option. Under the other analysis (7% discount rate), though, only one rule option (10 ppt PFOA and 10 ppt PFOS) has net benefits. EPA should recognize the significance of these conflicting results, in conjunction with the concerns regarding the cost analysis, prior to finalizing any rule.

5. The use of the Standard Monitoring Framework (SMF) for PFAS is appropriate where a running annual average (RAA) below one-half of the MCL is considered to be reliably below the MCL. However, the agency's proposed approach to require reporting results below the practical quantification level (PQL) to calculate the RAA for reduced monitoring is inappropriate and will cause equity issues with respect to access to high quality laboratories. This will lead systems with less financial capacity to have more stringent monitoring requirements. EPA should move forward with the SMF, where all results below the PQL are considered 0 ppt.

6. The affordability assessment relies on dated, inaccurate data and an approach that fails to capture affordability challenges for many communities. The affordability analysis should be updated to more accurately reflect household affordability and anticipated challenges for lower-income populations.

As further detailed below, unless the issues outline in this letter are addressed, AWWA is concerned that any final rule would be legally vulnerable for not complying with the SDWA and the Administrative Procedure Act (APA).

[1] Under 42 U.S.C. 300g–1(b)(10), the Administrator "may allow up to 2 additional years to comply with a maximum contaminant level if…additional time is necessary for capital improvements" (104th Congress, 1996).

3. Implementation Challenges

Public water systems subject to the proposal will need to comply within three years unless a two-year extension is provided by state primacy agencies or the Administrator. An estimated 67,000 water systems will need to perform initial monitoring at nearly 90,000 entry points used by water systems. Additionally, upwards of 4,300 water systems will need to take action to address PFAS levels above the maximum contaminant levels (MCLs) and continue to conduct quarterly sampling, according to the EPA's analysis. For water systems that need to install advanced treatment facilities for PFAS, a myriad of challenges will delay the implementation timeline for each system and will impact costs to implement these facilities.

Simply put, the current implementation timeline will cause the final rule to be infeasible, and therefore conflicts with the SDWA. When EPA establishes an MCL, the combination of technology, treatment techniques, or other means required to meet the level must not be more stringent than feasible.[2] The SDWA defines "feasible" to mean "feasible with the use of the best technology, treatment techniques and other means which . . . are available (taking cost into consideration)."[3] The tight timeline for implementation here would render the "the combination of technology, treatment techniques, or other means" infeasible because of the additional costs and implementation considerations.

As required by the SDWA, AWWA encourages EPA to consider how these implementation challenges will impact not only the compliance cost of the rule, but the feasibility of the rule to be implemented on the standard timeline provided by the SDWA. These challenges are further detailed below.

Laboratory Capacity is Lagging Behind Demand

As noted above, more than 67,000 water systems will be driven to comply with the initial monitoring requirements to determine their PFAS levels. Given the timeline of the rule, as described in the proposed rule preamble and reflected in the proposed rule text, water systems that may not leverage previously collected PFAS sampling data, will need to perform initial monitoring during the 12-months immediately following the rule's promulgation. The product of this surge in water monitoring sampling will require laboratories to process more than 220,000 water samples being collected by these systems.

This is in addition to ongoing monitoring activities by the water sector that include compliance monitoring for systems subject to state drinking water standards, performance testing by systems with treatment facilities, and samples to support pilot testing by systems investigating and designing new treatment facilities. Examples of sampling programs for testing new treatment facilities are laid out in detail in AWWA's "Drinking Water Treatment Selection Guide for PFAS" (AWWA, 2020a). This is also in addition to sampling being performed outside of the water sector for environmental investigations and the implementation of recent actions for effluent discharges, which will largely rely on the same laboratories (EPA, 2022d; EPA, 2022e). By comparison, approximately 20,000 water samples will be processed annually as part of the UCMR 5 program. Initial monitoring requirements will increase the demand for laboratory capacity by a factor of more than 11.

Over the past few years, the demand for the analysis of samples has continued to grow and has outpaced the increase in laboratory capacity. Water systems are currently reporting sampling challenges like longer processing and turnaround times, higher analytical costs, and less reliable reporting data

[2] 42 U.S.C. 300g-1(b)(B)(5)(B)(ii).
[3] 42 U.S.C. 300g-1(b)(B)(4)(D).

quality. The surge of sampling activity, especially with an emphasis on lower reporting levels, will further strain the existing laboratory capacity. EPA will therefore create unavoidable compliance risks for public water systems unless it extends the implementation timeline to the maximum extent possible.

Securing Financing is a Slow Process

A crucial step for installing capital improvement projects is to identify and secure a source of financing. In announcing this rule, EPA highlighted the funds through the Bipartisan Infrastructure Law (BIL) and the Drinking Water State Revolving Fund (DWSRF). Financing is also available through the Water Infrastructure Finance and Innovation Act (WIFIA) program. These programs provide an avenue for water systems to finance new treatment facilities, but these programs are known be time consuming and sometimes take several years to acquire approval, which is in addition to the time to get to a project design that can be reviewed. These programs may also impose additional requirements for funding to be approved that may limit procurement options and costs. Whether a system utilizes the DWSRF, WIFIA, or through the market, the process may still be slow and will be independent of the typical financial planning process of developing a Capital Improvements Program. Planning through this program helps to assure that capital improvements are staged in a way that minimizes water rate impacts by staggering major investments within a community's water infrastructure.

Simultaneous Compliance will Slow Down Implementation

The installation of new water treatment facilities requires sufficient planning to ensure that bringing a plant into compliance with a new rule does not cause non-compliance with existing regulations. For most water systems in the U.S., the installation of PFAS treatment facilities will create challenges for simultaneous compliance with existing drinking water rules. Each of the best available technologies for PFAS will have impacts on the finished drinking water and may require post-treatment to avoid negative impacts. For example, the use of reverse osmosis (RO) and anion exchange (IX) treatment can increase the corrosivity of water impacting the potential for lead release into drinking water at homes. Granular activated carbon (GAC) has been known to contribute to distribution system nitrification. These impacts can be mitigated, but mitigation requires adequate evaluation.

One such example where simultaneous compliance concerns will delay the implementation of new drinking water treatment is the requirements Lead and Copper Rule Revisions (LCRR) (EPA, 2021d). When systems determine that mitigation is needed to comply with any new PFAS standards – either through new treatment or a change in the water supply source – they will need to comply with the LCRR requirements, which could include a lengthy process of analysis and subsequent studies to obtain approval from their primacy agency. The LCRR corrosion control studies and subsequent actions could take years to achieve. When complying with the LCRR leads to significant changes in corrosion control treatment (CCT) after the rule's administrative procedures are following, a system must have time to: (i) prepare the distribution system and customers for the transition, (ii) shift corrosion control practice at a pace that does not lead to water quality concerns, and (iii) simultaneously install the required PFAS treatment or water supply option. This will have a significant impact on the system's ability to install PFAS treatment within three to five years of any final PFAS rule and impact the cost of implementing new treatment for PFAS.

Most Systems will Need to Perform Pilot Testing

Finally, another important step in installing PFAS treatment facilities is pilot testing. While GAC, IX, and RO have been documented as being capable of removing PFAS effectively, they still require a sufficient level of pilot testing. Pilot testing typically takes at least six to nine months to complete, and costs vary but include the rental cost of equipment, engineering and other technical support, and appropriate monitoring and sample analysis costs. Bench scale testing can also be useful and is less costly for some systems.

The are a number of important goals associated with pilot and bench-scale testing are 3-fold: (i) demonstration of PFAS removal efficacy, (ii) characterizing pre- and post-treatment needs, and (iii) optimal treatment technology selection, (iv) confirmation of design and operational parameters, and (v) estimation of capital, operations and lifecycle costs (AWWA, 2020a). It is anticipated that most of the water systems that must install treatment to meet PFAS MCLs will need to perform pilot testing, especially given the permitting requirements to comply with the LCRR, as discussed above. While pilot testing may not seem appropriate for smaller systems, it is similarly vital for these systems to ensure that the expense of capital for a new long-term treatment facility is both cost-effective and appropriately designed to protect public health from secondary water quality changes. The potential cost to small systems if new treatment facilities fail to operate as intended can be severe, given the cost of identifying and implementing solutions cannot be distributed across a large number of households, particularly after water rates are already rising to take on debt of the initial PFAS rule compliance solution.

Implementation will Further Strain the Supply Chain

When considering the costs and feasibility of the timeline and proposed rule, EPA must also take into account current supply chain issues. Water systems have been faced with a strained supply chain, which were worsened following the start of the COVID-19 Pandemic. This strain has led to increased purchasing costs, longer lead times for equipment or materials, and limitations on the products that are available. Lead times for key equipment (e.g., vessels, carbon or resin media, electrical components, etc.) have already increased to beyond twelve months, depending on the equipment and the degree of specialization it requires. Vessels, GAC media, and IX resin are not widely available from more than a few suppliers. Lead times for replacement GAC media are currently six months or more and for new customers the lead time is in the range of twelve to eighteen months. The lead time for GAC media will increase as a surge of new systems begin ordering GAC and suppliers will need to acquire media internationally (e.g., China and India) as the domestic market becomes more strained. Manufacturing of IX resin is currently not domestic given the safety concerns regarding the chemicals used in its production as demand for IX resin increases the supply chain is anticipated to strain as well. These issues are also impacting major ancillary equipment like electrical panels, motor control centers, etc.

Workforce Limitations will be Worsened

The water sector is currently working to overcome workforce challenges, which EPA must also recognize when considering the feasibility of the timeline and proposed approach. EPA estimates that one-third of the sector's workforce is eligible to retire within the next 10 years and water systems are facing challenges in recruiting, training, and retaining employees (EPA, 2023a).

These challenges are expected to be more severe as more than 4,300 water systems are driven to advanced technologies that require more specialized technical skills. To install and operate these facilities, water systems will need to hire or contract engineers, manufacturers and suppliers, construction crews, and skilled operators. These service providers are already in high demand and in short supply. The resulting imbalance is impacting labor and material costs, lead times for materials, turnaround times for services (e.g., engineering, laboratory analysis, construction). The installation of new treatment facilities will surge following rule's promulgation, which will further worsen workforce challenges for water systems.

The demand for highly skilled water treatment operators will increase due to this rule. Systems currently not using filtration for water treatment may need to meet additional operator certification requirements. While each state independently sets certification requirements for water treatment plant operators, it is anticipated that systems requiring to install GAC, IX, or RO will need to staff operators with more advanced certification. Water systems in the states of Virginia[4], California[5], Colorado[6], and Massachusetts[7], for example, will all see impacts to operator certification requirements as a result of new treatment systems for PFAS. This change will have a significant impact on systems with a limited local labor pool or limited financial capacity to attract skilled operators that will be needed to safely operate these advanced treatment systems. As EPA recognizes, systems must have staffing with appropriate qualifications to operate 24 hours a day, 365 days a year. Many systems will struggle to find qualified operators for adequate staffing.

State Primacy Agency Capacity

Because the typical timeline for the planning, design, permitting, and construction of a new drinking water treatment facility for PFAS may take up to and exceed 5 years, the standard compliance window of three years under SDWA will not be feasible. This is especially important given that these implementation challenges will drive the timeline up as systems begin competing for the same limited supply of sector resources. Under SDWA, water systems may request a two-year extension for compliance with MCLs if it is determined that additional time is necessary for capital improvements.

It is anticipated that the vast majority of water systems that need to install treatment for PFAS will need to request this two-year extension, which is typically provided at the discretion of the state primacy agencies. State primacy agencies are currently working to review lead service line inventories, preparing to implement the corrosion control treatment requirements of the LCRR, administering the DWSRF and additional projects accessing funds from the Bipartisan Infrastructure Law, and working to ensure and improve water system compliance with existing rules. As the surge in water system requests for a two-year extension begins, these agencies will be strained as they work to review and process these requests.

However, the Administrator also has the authority under SDWA to provide this extension and can do so as a part of the rule as opposed to being done so on a case-by-case basis. In order to prevent issuing a

[4] 18 Virginia Administrative Code 160-30-370 – Waterworks.
[5] California Code Regs. Tit. 22, § 64413.1. - Classification of Water Treatment Facilities.
[6] 5 Code of Colorado Regulations 1003-2 Regulation 100 - Water And Wastewater Facility Operators Certification Requirements: Sections 100.4 to 100.9.
[7] 310 Massachusetts Register 22.11B.

final rule that is infeasible due to the implementation timeline or otherwise violates the APA as arbitrary and capricious, AWWA recommends that the Administrator leverage this authority to increase the likelihood that all water systems can comply with the timeline of the rule and take adequate, effective steps towards mitigating PFAS levels in drinking water.

4. Analysis of Occurrence Data

The SDWA requires that EPA rely upon the best available public health information, including the occurrence database.[8] EPA must therefore ensure that the data on which it relies meets this standard. Because "best" is necessarily comparative, EPA must also provide sufficient explanation regarding the data selected and not selected as the basis of EPA's decision so that the public can meaningfully comment on the data selected as well as evaluate the data that EPA did not decide to rely upon.

According to the proposal, EPA applied a statistical modeling approach to characterize occurrence data for PFAS using a combination of both national occurrence monitoring data from the UCMR 3 and more recently collected state data. While EPA typically relies on nationally representative occurrence data from the UCMR program to drive decisions for NPDWRs, the agency previously noted an interest in using data collected by state monitoring programs given the UCMR 3 database's high reporting limits relative to the potential levels of health concern (EPA, 2021e). AWWA appreciates the agency's interest in advancing this rulemaking by leveraging more recently collected data using improved methods. A detailed understanding of contaminant occurrence in drinking water across the country is a key factor for developing not only regulatory determinations, but also drinking water standards. In review of this approach, several opportunities to improve the analysis were identified.

Application of the Bayesian Statistical Model

The engine of the occurrence analysis for the proposal is the Bayesian hierarchical statistical model (the Bayesian Model), that uses PFAS occurrence data from more recent monitoring programs to provide improved understanding of the UCMR 3 data below the reporting limits. Similarly, the occurrence analysis is the engine of the entire rulemaking, informing the EPA's understanding of the regulatory impacts of the rule. For this reason, it is imperative that the Bayesian Model be utilized appropriately and with statistical confidence.

While the Bayesian Model approach is sophisticated technically, it is an overly complex approach for characterizing national occurrence. Bayesian models can be useful in many applications but there are some key challenges that arise with the use of these models that make it non-optimal for regulatory applications. The key challenge is that the selection of priors and the posterior conditions is a very subjective decision that is subject to the discretion of the statistician that is crafting the model. As such, it is expected that the assumptions around these decisions are documented clearly and in detail. While EPA has provided a copy of the code used for the Bayesian Model, along with a recent publication about the model, there is a lack of a non-technical description of the agency's intended approach. The lack of a clear explanation of intent regarding the model's code leaves stakeholders unable to confirm that the code is accurately developed, and therefore unable to meaningfully comment on this aspect of the proposal.

[8] 42 U.S.C. § 300g-1(b)(1)(B)(ii)(II).

Additionally, EPA only provided a 60-day public comment period for review and analysis of the entire proposal, including the occurrence analysis. AWWA and numerous additional stakeholders requested an extension of the comment period to support more in-depth of the occurrence analysis and the rule more broadly but the agency declined this request. The comment period is inadequate for reviewing this model and EPA neglects to provide informative sensitivity analyses and to clearly present model assumptions and outputs. In order to fulfill its obligations under the APA, AWWA therefore requests that EPA provide this information during a supplemental comment period prior to finalizing any rule.

Use of Non-UCMR 3 Data

EPA's occurrence analysis relies on data from both UCMR 3 and state monitoring programs. AWWA supports the consideration of the more recently collected data from state monitoring programs to improve understanding of occurrence, but there are several concerns about the agency's use of this data and the degree of quality control. These issues are discussed in more detail in the following paragraphs. As previously noted, the SDWA requires that EPA rely upon the best available public health information, including the occurrence database.[9] EPA must also provide sufficient information and explanation regarding the data selected and not selected as the basis of EPA's decision so that the public can meaningfully comment on the data selected as well as evaluate the data that EPA did not decide to rely upon.

It is unclear how data was screened for inclusion as part of the analysis. Several state monitoring datasets are documented to have reporting thresholds far below what is considered reliable for a national occurrence analysis; for example, reporting thresholds below 1 ppt are indicated for several states including New Jersey, Massachusetts, and California. In other cases, states did not indicate the applicable reporting thresholds. While the proposal acknowledges these data quality issues, the agency nonetheless elected to utilize this data without quality control. As AWWA noted in 2020, EPA should supplement monitoring data from UCMR 3 with high quality occurrence data (AWWA, 2020b). It is recommended that EPA re-evaluate the non-UCMR 3 data that is being leveraged and ensure that monitoring results that are neither achievable using the robust methods approved by EPA nor representative of high-accuracy data should not be considered as part of this analysis.

Additionally, EPA has noted that the Bayesian Model's incorporation of state monitoring data excludes all non-UCMR 3 data that was collected by water systems that did not participate in UCMR 3. While EPA rationalizes this decision by highlighting that the UCMR 3 program was designed to collect data that is nationally representative, this approach fails to realize an opportunity to leverage the vast quantity of non-UCMR 3 data that is available. A project was conducted by Corona Environmental Consultants for AWWA that collected PFAS monitoring data from both UCMR 3 and state monitoring programs (Corona, 2021). This work successfully aggregated data from these programs from nearly 8,000 public water systems from across the country. Of these systems, 668 systems had participated in UCMR 3 and had more recent data available through state monitoring programs. Additionally, data was available for more than 3,100 water systems had participated in state monitoring programs but not UCMR 3. More effective inclusion of this data would expand the more recent data set for PFAS occurrence by a factor of nearly 5, which could improve our understanding of occurrence for smaller systems significantly.

[9] 42 U.S.C. § 300g-1(b)(1)(B)(ii)(II).

It is reasonable that the EPA is interested in leveraging the non-UCMR 3 data in a way that is nationally representative, but there are two aspects of the proposed approach that require a more detailed review. First, it is not clear why EPA is willing to leverage an overly complicated Bayesian Model to assess occurrence but at the same time is unwilling to develop a statistical approach to incorporating the additional non-UCMR 3 data in a manner that is nationally representative. EPA has existing experience under the UCMR program selecting small water systems for participation in the UCMR program that will be nationally representative. It is not apparent that EPA considered or attempted to leverage the significantly sized dataset of non-UCMR 3 data more effectively, especially to improve the occurrence analysis for smaller systems. In order to fulfill its obligations under the SDWA, EPA must likewise use a statistical approach to the smaller water systems in order to make better use of this dataset.

Additionally, it is likely that the non-UCMR3 data that EPA used for the Bayesian Model is biased and not nationally representative. Consequently the data EPA used is likely not the best available public health information to support the rulemaking. Bias is present in the data collected after UCMR 3 is correlated to states with elevated concerns about statewide PFAS contamination following UCMR 3 detections and improved understanding of likely sources of PFAS in the state. While EPA notes that non-UCMR 3 data collected by systems that did not participate in UCMR 3 should be excluded because it may not be nationally representative, the agency did not determine whether the systems that were included are nationally representative. Unless EPA addresses its inconsistent treatment of available data before issuing any final rule, it risks violating both the SDWA and the APA.

Finally, the occurrence analysis was improperly used to project a probabilistic distribution of PFAS levels across all water systems in violation of the SDWA and APA. This approach may be appropriate for systems where there is not available data as that actual data is the best available data. However, in taking this approach EPA cannot ignore data for specific system PFAS levels. These systems, that have previously collected data, should be captured in the occurrence analysis based on their previously collected data. Similarly, EPA should reflect actual data when available in its EPA's occurrence analysis of PFAS and total organic carbon (TOC). Besides ensuring that the costs for these systems are accurately reflected, this will ensure that any unique relationships that may exist between TOC occurrence and PFAS occurrence will be captured.

The SDWA explicitly provides a mechanism for EPA to obtain nationally representative occurrence data for contaminants in drinking water by requiring EPA issue a new list of unregulated contaminants to be monitored in drinking water every five years.[10] This list is known as the UCMR. The UCMR serves to better inform regulatory determinations, as contaminants are evaluated based on health effects and occurrence information, and EPA has historically relied on the UCMR process to collect occurrence data on contaminants to support a determination on whether to regulate contaminants. There are times when more recent or robust data may be available outside of UCMR collection,[11] and in such cases, EPA can appropriately rely on a combination of UCMR and non-UCMR data when available, so long as in doing so it provides a reasoned explanation for its approach and ensures that it is relying on the best available data for national occurrence. EPA has not done so here and must revise its data and provide greater transparency into its data in order to fulfill its obligations under the SDWA.

[10] 42 U.S.C. § 300g-1(b)(1)(B).
[11] 87 Fed. Reg. 68060, 68062 (November 14, 2022).

Inclusion of Data from Fifth Unregulated Contaminant Monitoring Rule

This year, drinking water monitoring for PFAS under the Fifth Unregulated Contaminant Monitoring Rule (UCMR 5) began. UCMR 5 is expected to be the most comprehensive occurrence dataset for PFAS collected to-date. Indeed Congress took the additional step of explicitly instructed EPA to include these substances in UCMR 5 as part of the National Defense Authorization Act for Fiscal Year 2020 (Congress, 2019). In accordance with UCMR 5, more than 10,300 systems will monitor for 29 PFAS using EPA Methods 533 and 537.1 using single-digit minimum reporting limits. While a complete dataset will not be compiled until 2026, more than 3,500 water systems are actively collecting monitoring data this year. By the end of summer this year, EPA will have at least one sample result from these 3,500 water systems. Previous research has shown that preliminary data collected from UCMR monitoring provides accurate insights about occurrence when compared to the complete dataset (Eaton et al, 2018).

UCMR 5 data collected during the first half of 2023 would represent a significant increase in the available data that is nationally representative. Specifically, this dataset will provide information on nearly 70% of the number of water systems that are typically represented by a UCMR program. While this data would not be a full UCMR sampling program, it would significantly expand the universe of nationally representative data at method reporting levels deemed appropriate in the proposed rule. As the goal of the UCMR 5 program is to inform EPA on the occurrence of PFAS, EPA can ill afford to ignore these data, which would be of a much higher quality and value than its current Bayesian Model approach.

The UCMR program is designed to collect national occurrence data on contaminants not currently subject to NPDWRs, and EPA "require[ed] collection of data under UCMR 5 to inform EPA regulatory determinations and risk-management decisions" (EPA, 2021c). Given that Congress explicitly instructed EPA to include PFAS chemical in UCMR 5, Congress clearly intended for the national occurrence date resulting from UCMR 5 to inform EPA's regulatory determinations about these substances. This strongly suggests that EPA should wait to take final regulatory action on these substances until all UCMR 5 data has been collected so that that its decisions can be fully informed with the best available information. This is particularly true given that the SDWA's anti-backsliding provisions will require continued regulation of these substances once EPA has issued a final NPDWR: EPA must make the most informed decision possible at this stage to fulfill its statutory obligations and prevent unnecessary and unjustified regulations.[12]

But even if EPA does not wait until all UCMR-5 data has been collected, it must at the very least incorporate and prioritize data already provided to the agency under UCMR-5 in making regulatory decisions under this proposal. Given that much of the UCMR 5 data has already been collected, EPA cannot meet the SDWA's directive to rely on the best available public health information without taking into account this most current and comprehensive set of data.

[12] *See* 42 U.S.C. § 300g-1(b)(9) ("Any revision of a national primary drinking water regulation shall be promulgated in accordance with this section, except that each revision shall maintain, or provide for greater, protection of the health of persons.").

Distinguishing Between PFAS Detections and Levels of Health Risk Concern

The SDWA only allows EPA to regulate a substance when "the contaminant is known to occur or there is a substantial likelihood that the contaminant will occur in public water systems with a frequency *and at levels of public health concern*."[13] The agency's discussion about the occurrence of PFAS in drinking water frequently discusses the likelihood of detections of PFAS. While this is useful, it would be more relevant, and therefore beneficial, for EPA to provide information about PFAS occurrence at levels closer to the relevant health risk levels, particularly the proposed MCLGs or MCLs. While occurrence at any level is useful to understand, it is misleading to the public and it is important that occurrence be contextualized with the relevant levels of health concern, given that that is the proper statutory focus under the SDWA. A detection of PFBS at 5 ppt, for example, represents a level that is less than 0.25% of the EPA's lifetime health advisory level (HAL). In comparison, a detection of PFHxS at 5 ppt represents a level that is 55% of the EPA's proposed health-based water concentration (HBWC). Put plainly, a detection of PFBS represents a much different level of risk than a detection of PFHxS. For this reason, EPA should ensure that the occurrence analysis provides context on PFAS occurrence that is more useful than detection.

Providing Transparency of Occurrence Analysis Outputs

While it has been previous practice of EPA to depict the conclusions of its occurrence analysis by presenting the number of water systems (in addition to the population) impacted by the proposed rule and the rule options. EPA failed to follow that practice here and did so without providing a proper explanation for this change in its analytical approach, as required by the APA. Instead, EPA provided a breakdown of the population impacted along with a more limited set of information about the small systems that are impacted.

AWWA previously inquired about additional information on Bayesian Model in Fall 2022 and received a series of statistical outputs that did not assist AWWA in an understanding the PFAS occurrence at systems. Following the rule's publication to the Federal Register AWWA discussed the absence of this data in the Docket with EPA staff during a conference call; in discussion with staff, EPA clarified that the data, in their entirety, was not available as part of the supporting information provided for this rulemaking.[14] AWWA made a written request for this data following this conference call and has not received the completed information (Moody, 2023).

It is important that relevant occurrence information underpinning a rulemaking analysis be made available for public comment and included in the record as part of any final PFAS rule, and for all future proposals for national primary drinking water regulations, so that the public can understand the overall impact of the rule on communities and to confirm that the benefits and costs attributed to the rule are accurate.

[13] 42 U.S.C. § 300g–1 (b)(1)(A)(ii).

[14] Conference call on March 28, 2023, between AWWA and EPA staff.

Summary of Recommendations for Improving Occurrence Analysis

In order to comply with the requirements of the SDWA and APA, AWWA makes the following recommendations to improve quality and transparency of the occurrence analysis:

1. The agency should provide clearer and more transparent information on the intended approach of the Bayesian Model, including information on the model outputs with respect to the number and type (e.g., system size, source) of water systems impacted by the various regulatory options.
2. The agency's consideration of non-UCMR 3 data needs additional quality control to exclude system data that does not meet the necessary data quality for national representation, such as reported results that are below nationally reliable reporting limits, as recognized by EPA.
3. Non-UCMR 3 data should be leveraged more effectively. Specifically:
    a. The agency should leverage existing methodologies used to by the UCMR program to incorporate data from systems beyond the scope of UCMR 3 as part of the Bayesian Model in a way that maintains the national representation of the data and provides additional confidence.
    b. If the agency determines that there are insufficient resources within the agency to do this, the non-UCMR 3 data that is excluded from the Bayesian Model should be used to evaluate the Model outputs.
4. The non-UCMR 3 data that is included in the Bayesian Model and considered to be nationally representative should be evaluated and EPA should substantiate the basis for its inclusion.
5. The agency should consider a 2-tier approach that relies on best-available, system specific data on PFAS levels and TOC levels and relies on a probabilistic distribution approach to the remaining systems without known PFAS and TOC levels.
6. EPA should not finalize the occurrence analysis without considering the availability of high-quality, nationally representative data from the UCMR 5 program to either improve the existing occurrence analysis or replace the analysis.

5. Preliminary Regulatory Determinations

The proposal includes preliminary regulatory determinations for four PFAS (and their mixture) concurrently with a proposed drinking water regulation of these compounds. AWWA supports the agency's interest in looking at PFAS beyond PFOA and PFOS for potential action and has previously recommended that the agency do so by applying adequate resources to fill data gaps (AWWA, 2020b).

Under the SDWA, EPA may only issue a NPDWR for a contaminant that is known to occur or there is a substantial likelihood that it will occur in public water systems at a level of public health concern.[15] In the preamble to the proposal, the agency notes that there is not a "bright-line threshold for occurrence in drinking water that triggers whether a contaminant is of public health concern". AWWA agrees that SDWA does not define a "bright-line threshold" that would define that a contaminant is of public health

[15] 42 U.S.C. § 300g–1 (b)(1)(A)(ii).

concern but given the statutory focus on the "adverse effect on the health of persons"[16] it would be arbitrary and capricious and conflict with the SDWA if EPA did not use the level of adverse health effect to represent the level at which a contaminant starts to be considered a public health concern. AWWA also notes that EPA should consider its past practices for determining whether a contaminant reaches a level of public health concern and ensure that its approach in any final rule is consistent with past practice or that it provides a reasoned explanation for any deviation from past practice. AWWA offers recommendations for each of these options, with this and other aspects in consideration. AWWA further notes that the best available health information indicates that a negative determination is appropriate for PFNA, HFPO-DA, and PFBS at this time.

PFHxS

As part of the proposal, EPA has developed a HBWC using minimal risk levels that were developed by the Agency for Toxic Substances and Disease Registry (ATSDR) (ATSDR, 2021). AWWA supports the use of the proposed HBWC as a screening level for PFHxS in support of the regulatory determination as EPA's Integrated Risk Information System (IRIS) program works towards completing its health assessment (EPA, 2023b).

Occurrence data for PFHxS is available not only through the UCMR 3 program but also as part of numerous state monitoring programs. Data is also currently being collected through the UCMR 5 program by 3,500 systems this year and more than 10,000 systems by the end of 2025 (EPA, 2021c). In review of the UCMR 3 data that is currently available, approximately 1.1% of water systems detected PFHxS at 30 ppt, more than 3 times higher than the proposed HBWC of 9.0 ppt. Additionally, data from California, Colorado, Pennsylvania, Vermont and Ohio show a similar trend of occurrence at levels above the proposed HBWCs (California Water Boards, 2023; CDPHE, 2023; Ohio EPA, 2023; PADEP, 2023; VTDEC, 2023). In review of the available occurrence data in comparison with the EPA's proposed HBWC, AWWA agrees that there is evidence that PFHxS occurs in drinking water at potential levels of concern.

HFPO-DA

The UCMR 3 program did not include monitoring for HFPO-DA. Therefore, EPA is not able to determine the national occurrence of the chemical. While some states have conducted monitoring for HFPO-DA, these states provide only a limited understanding of national occurrence.

In review of the EPA's analysis, HFPO-DA monitoring was conducted in only 16 states and the data does not provide sufficient evidence that there is national occurrence of HFPO-DA. In two states, North Carolina and Alabama, sampling data was available, but the extent of the program was unknown and so statewide occurrence levels can not be determined. In eight states, HFPO-DA was not detected in any systems and in another three states there was less than 0.2% of systems with detections of HFPO-DA, let at levels above the EPA's lifetime health advisory level of 10 ppt (EPA, 2022f).

The only state with a significant number of detections of HFPO-DA at systems was Kentucky. A total of 81 systems were sampled across the state. Of these 81 systems, eleven detected HFPO-DA. An in-depth review of this data shows that all of these systems, except one, rely on the Ohio River as a water supply source. The last system relies on the Ohio River Alluvium. This data suggests that HFPO-DA contamination in Kentucky is not widespread but rather linked directly to recent releases directly to the

[16] 42 U.S.C. § 300g–1 (b)(1)(A)(i).

Ohio River. Specifically, it is anticipated that the Ohio River, and these systems, have been impacted by discharges of HFPO-DA from the Washington Works PFAS manufacturing plant in Parkersburg, West Virginia.

The Washington Works plant has long been a center for discussions on PFAS contamination. However, the EPA recently took landmark action against this plant for violations of the Clean Water Act related to discharges of HFPO-DA to the Ohio River (EPA, 2023c). With this action, it is anticipated that HFPO-DA levels in the Ohio River will drop, which will lead to a reduction in contamination of affected systems. Similar action has been taken by the North Carolina Department of Environmental Quality (NCDEQ) against the Fayetteville Works facility along the Cape Fear River. In 2019 NCDEQ issued a consent order that required the facility to begin taking mitigative measures against the release and contamination of HFPO-DA in the area surrounding the facility. Following EPA's publication of the lifetime health advisory level in 2022, this consent order was updated and now will limit discharges to a maximum of 10 ppt HFPO-DA. This action will reduce HFPO-DA contamination in the Cape Fear River.

Data is available on the production, use, and release of HFPO-DA from the EPA's supporting documentation and shows that HFPO-DA was released by five facilities in five states. According to the most recent TRI program data for HFPO-DA, 72% of the total HFPO-DA and its ammonium salt released was from the Fayetteville Works facility in North Carolina; the Washington Works facility in West Virginia accounted for 5.7% to the total releases (EPA, 2023d). Given that both of these facilities are reducing releases, as discussed above, this will reduce the total release of PFAS by as much as 77.7% from these two facilities alone. Further reductions are anticipated following the promulgation of the ELGs for manufacturers and metal finishers under the CWA, which has been identified by EPA as a part of the Effluent Guidelines Program Plan 15 (EPA, 2021b). Similar reductions may be anticipated as EPA and states work towards addressing PFAS as part of the National Pollutant Discharge Elimination System (NPDES), as directed by the agency in April and December of 2022 (EPA, 2022d; EPA, 2022e).

Overall, UCMR data is not currently available for HFPO-DA and the available state data is not sufficient to determine the national occurrence of HFPO-DA in drinking water. Furthermore, the limited occurrence observed by state monitoring programs and the limited extent of production shown by information from the TRI program data is not suggestive of a substantial likely of HFPO-DA occurrence in drinking water with a frequency and at levels of public health concern. Instead, the available evidence indicates that a negative determination is appropriate for HFPO-DA. However, given that the EPA is currently collecting occurrence data for HFPO-DA in drinking water as part of the UCMR 5 program, EPA could consider re-issuing a preliminary determination for HFPO-DA following the completion of this program. This approach would ensure that the best available occurrence data is used.

PFNA

As with PFHxS, EPA also developed a HBWC for PFNA using data from ATSDR (ATSDR, 2021). AWWA supports the use of the proposed HBWC as a screening level for PFNA to guide this determination in the absence of a completed IRIS program health assessment (EPA, 2023b).

Occurrence data for PFNA is available not only through the UCMR 3 program but also as part of numerous state monitoring programs. Data is also currently being collected through the UCMR 5 program by 3,500 systems this year and more than 10,000 systems by the end of 2025 (EPA, 2021c). In review of the UCMR 3 data that is currently available, less than 0.3% of water systems detected PFNA at

20 ppt (twice the level of the proposed HBWC of 10 ppt). Data from state monitoring programs showed similarly extremely low occurrence of PFNA in drinking water. Vermont, for example, required sampling of 1,794 water systems across the state and PFNA was detected above 10 ppt in only 12 systems, or 0.7% of systems (VTDEC, 2023). California monitoring, for example, found that 95% of samples with detections were below 3.2 ppt (California Water Boards, 2023). Monitoring data from Pennsylvania Department of Environmental Protection showed a maximum PFNA concentration of 14 ppt in the state, with a median PFNA level of 5.6 ppt (PADEP, 2023). Data from showed a similar trend of low to minimal occurrence at the HBWC (CDPHE, 2023).

The available data on the production, use, and release of PFNA from the EPA's occurrence analysis indicates that there are not significant sources of PFNA in the United States. While this could be due to inefficiencies in the reporting requirements under the EPA's authorities of the Emergency Planning and Community Right To Know Act's Toxics Release Inventory (TRI) program, the agency must rely on the best available data (AWWA, 2023). EPA recently proposed rules that will require more improved data reporting on PFAS production, use, and release in the following years (EPA, 2022b).

In review of the available UCMR 3 data, state monitoring data, and manufacturing data for PFNA, the best available evidence does not suggest that there is a substantial likelihood of PFNA occurrence in drinking water with a frequency and at levels of public health concern. Based on the available evidence, and the SDWA statutory criteria, a negative determination is most appropriate for PFNA. However, given that the EPA is currently collecting occurrence data for PFNA in drinking water as part of the UCMR 5 program, EPA could consider re-issuing a preliminary determination for PFNA following the completion of this program. This approach would ensure that the best available data is utilized, not only occurrence data but also a forthcoming health assessment for PFNA under IRIS program (EPA, 2023b).

PFBS

The method reporting limits of the UCMR 3 program for PFBS provide sufficient clarity on occurrence at levels far below levels of health concern. For example, EPA's health advisory level is 2,000 ppt for PFBS and was only detected in less than 0.2% of systems above the reporting limit of 90 ppt under UCMR3 (EPA, 2022g) . Data from the state of Pennsylvania showed that the maximum PFBS level in drinking water was 13.0 ppt, with a median detected concentration of 4.2 ppt (PADEP, 2023). California monitoring data found that the maximum concentration across the state did not exceed 120 ppt and 95% of systems detected PFBS had levels below 15 ppt (California Water Boards, 2023). Monitoring data from several other states, including Ohio, Colorado, and Vermont show a similar trend (CDPHE, 2023; Ohio EPA, 2023; VTDEC, 2023).

Additionally, the available data on the production, use, and release of PFBS from the EPA's occurrence analysis indicates that there are not significant sources of PFBS in United States. While this could be due to inefficiencies in the reporting requirements of the TRI Program, the agency must rely on the available data. EPA recently proposed rules that will require more improved data reporting on PFAS production, use, and release in the following years.

In review of the available occurrence data, both from UCMR 3 and from state monitoring programs, it is apparent that PFBS does not occur in public water systems with a frequency, and at levels, of public health concern. The evidence from the production, use and release data that is available supports this conclusion. Therefore, at this time the best available public health information does not support a

determination to regulate PFBS under the SDWA. In contrast, there is strong evidence that a negative determination is appropriate for PFBS and AWWA recommends that a negative determination be issued. EPA can revise its determination as to PFBS in the future, if new data indicates that such a determination is warranted.

Alternatively, given that the EPA is currently collecting occurrence data for PFBS in drinking water as part of the UCMR 5 program, EPA could consider re-issuing a preliminary determination for PFBS following the completion of this program.

PFHxS, PFNA, HFPO-DA, and PFBS as a Mixture

As part of the proposal, EPA also sought public comment on the preliminary regulatory determination for PFHxS, PFNA, HFPO-DA, and PFBS as a mixture. Specifically, EPA highlights that Section 1401(6) of SDWA defines the term contaminant to mean "any physical, chemical or biological or radiological substance or matter in water" and therefore a mixture of two or more "contaminants" qualifies as a "contaminant" because the mixture itself is "any physical, chemical or biological or radiological substance or matter in water."

AWWA appreciates the agency's interest in addressing additional PFAS beyond PFOA and PFOS. However, the proposed approach to address these additional PFAS through a preliminary determination that these PFAS as a mixture meet the definition of being a single "contaminant" under SDWA is not appropriately supported. As recognized by the TSCA, any mixture is not considered a chemical substance, instead a mixture of contaminants formed either naturally or through a chemical formulation process may be considered a chemical substance (EPA, 2023e). EPA's "Supplemental Guidance for Conducting Health Risk Assessment of Chemical Mixtures", which EPA cites in crafting the risk assessment framework for the proposal, specifically indicates that opportunities to infer hazard for a mixture must be from a **"sufficiently similar mixture"** (EPA, 1986)**.** The guidance goes on further to note that a mixture is sufficiently similar when the **"components and respective portions exist in approximately the same pattern"** (EPA, 1986).

The occurrence analysis provided by EPA does not demonstrate that PFHxS, PFNA, HFPO-DA, and PFBS can be grouped together as a mixture within a manner consistent with well-established agency guidance. First, there is a complete lack of national occurrence data for HFPO-DA and the data for PFBS shows a significant lack of occurrence in drinking water. Furthermore, data for PFNA shows a very low level of occurrence on its own, let alone with the other three PFAS. EPA's groupwise occurrence provides neither a clear nor transparent characterization of occurrence, nor co-occurrence, of PFNA, PFHxS, PFBS, and HFPO-DA. The supporting documentation fails to illustrate a pattern of co-occurrence of these four compounds; in fact, most of the information on co-occurrence of these compounds is relative to PFOA and PFOS. Given that the determination is for PFHxS, PFNA, HFPO-DA, and PFBS the co-occurrence of these individual compounds with PFOA and PFOS does not demonstrate the other four compounds co-occur.

Furthermore, the information that is provided does not demonstrate occurrence of PFHxS, PFNA, HFPO-DA, and PFBS at or above levels of potential health risk. Instead, occurrence is consistently described as a function of detection. Unfortunately, this is also not useful in supporting a preliminary determination. Detections are not equivalent to potential risk, which is most easily demonstrated in comparing the meaning of a detection for PFBS and PFHxS. While a detection of PFHxS at 10 ppt is 100% of the

proposed HBWC while a detection of PFBS at 10 ppt is only 0.5% of the proposed HBWC. Detection of PFBS is likely to bias these results and represents a significantly different level of public health concern in comparison to a detection with other PFAS.

To further investigate co-occurrence for the PFAS, AWWA conducted an analysis of PFAS occurrence data that was collected from nearly 8,000 water systems by Corona Environmental Consulting (Corona, 2021). The results of this analysis are shown in the table. While some systems may detect more than one of these PFAS, the occurrence of these PFAS together at levels above the HBWC are much more limited, if at all.

Table 4-1: Co-Occurrence of PFBS, PFHxS, and PFNA in Drinking Water (N=7,989) (Corona, 2021)

	Number of Systems with Detections (% of Systems)	Number of Systems with PFAS Level(s) Exceeding HBWCs (% of Systems)
PFHxS & PFNA	605(7.5%)	32 (0.4%)
PFBS & PFHxS	765 (%)	0 (0%)
PFBS & PFNA	561 (%)	0 (0%)
PFBS, PFHxS & PFNA	551 (%)	0 (0%)

While there are scientific studies evaluating the hypothesis that exposure to multiple PFAS may lead to adverse health effects, the proposal and its supporting documentation do not substantiate a preliminary determination nor a determination to regulate the mixture of PFHxS, PFNA, HFPO-DA, and PFBS. There is a lack of information showing co-occurrence and, therefore, co-exposure to these compounds.

Equally importantly, EPA's proposed approach to using the general hazard index is significantly flawed and is not supported by federal agency guidance nor recommendations from the Science Advisory Board (SAB) (ATSDR, 2018; ATSDR, 2022; EPA, 1986; EPA, 2000; SAB, 2022), which collectively recommend that a common health outcome should be used as the basis for a hazard index in this context.

AWWA does not support the preliminary determination of PFHxS, PFNA, HFPO-DA, and PFBS as a mixture as the statutory factors under the SDWA to support a determination are not present. While EPA claims that mixtures of these PFAS may co-occur and represent a combined risk, the supporting information fails to create a sufficient record that this is the case. This is true both for evidence of occurrence and demonstration that these PFAS pose a combined risk. This lack of toxicological support for this approach is apparent in the agency's proposed approach to use a hazard index for these compounds through a methodology that is contrary to federal agency guidance. The preliminary determination for PFHxS, PFNA, HFPO-DA, and PFBS as a mixture is not sufficiently supported and the information for three of these compounds suggests a negative determination is most appropriate.

AWWA recommends that if EPA is interested in addressing additional PFAS through a regulatory determination for a mixture of PFAS, that EPA reconsider their approach to addressing PFAS as a mixture and delay re-issuing a preliminary determination until after the data collection activities for UCMR 5 are complete. Delaying this action would also ensure that EPA may consider health assessments for additional PFAS, which are currently in development (EPA, 2023b).

6. Proposed Maximum Contaminant Level Goals

The derivation of a science-based maximum contaminant level goal (MCLG) is crucial because it means that it is both protective of public health and can transparently be communicated to inform decision-making for the public. EPA did not use the best available science in proposing the MCLGs (and MCLs) for these substances as required by SDWA. EPA must also ensure that the underlying science is review by the EPA SAB. Because EPA has not demonstrated that PFNA, PFHxS, HFPO-DA, and PFBS warrant regulation under SDWA, it cannot finalize the proposed MCL and MCLG. AWWA offers the following specific comments on the proposed derivation of the MCLGs for PFOA, PFOS, and the mixture of PFHxS, PFNA, PFBS, and HFPO-DA.

PFOA

According to the proposal, EPA is proposing an MCLG of 0 ppt (zero) for PFOA based on a determination that PFOA is likely to be carcinogenic. AWWA has previously reviewed the EPA's determination that PFOA is carcinogenic and provided comments (AWWA, 2021a). Key aspects of those comments are shown below and can be found in more detail in Appendix A.

1. EPA cites Shearer et al (2021) as a key study showing that PFOA may be carcinogenic. This study may not be suitable as evidence to support this determination given that the study duration spanned less than 18 years. Given the half-life of PFOA, it is unlikely to accurately portray the exposure relevant to the development of kidney cancer.

2. In epidemiological studies of higher exposures there has been inconsistent evidence of increased cases of kidney cancer. For example, epidemiological studies of residents exposed to PFOA and other PFAS in contaminated drinking water have reported modest increases whereas occupational cohorts have shown increased and decreased risk of kidney disease, despite higher exposure and longer study durations.

If EPA moves forward with a conclusion that PFOA is carcinogenic, AWWA agrees that the appropriate MCLG for a carcinogen is 0 ppt (zero).

PFOS

As with PFOA, EPA is proposing to establish a MCLG of zero (0 ppt) for PFOS following a determination that PFOS is suggestive to be carcinogenic. As noted in comments in 2021, AWWA supports this determination (AWWA, 2021a). If EPA moves forward with a conclusion that PFOA is carcinogenic, AWWA agrees that the appropriate MCLG for a carcinogen is 0 ppt (zero).

Combined MCLG for PFHxS, PFNA, HFPO-DA, and PFBS

According to the proposal, EPA is proposing to establish a combined MCLG for four PFAS set at a hazard index of 1.0. In proposing this MCLG, EPA is making several key scientific determinations to support this decision:

1. PFNA, PFHxS, HFPO-DA, and PFBS are likely to co-occur in water in a way that is a "sufficiently similar mixture",

2. Co-exposure to a mixture of PFNA, PFHxS, HFPO-DA, and PFBS can lead to an aggregate health effect as a result of dose additivity, and

3. The dose additivity of PFAS can be applied through the hazard index with dissimilar health effects, or outcomes.

AWWA contracted with Ramboll Consulting U.S. to assist in reviewing the EPA's approach and to offer detailed recommendations to improve this work. Ramboll has provided a detailed letter with recommendations, which is included as part of Appendix A.

AWWA supports the agency's interest in taking a public health protective stance on PFAS. It cannot do so based on the assumption of dose additivity without sufficient evidence. There are numerous concerns regarding the agency's determination that these compounds co-occur and that their co-exposure has a dose-additive effect on dissimilar outcomes. Based on the information that EPA has currently provided and relied upon, EPA has not met the statutory or scientific requirements to make a positive regulatory determination for these substances individually or as a mixture.

First, AWWA notes that EPA's novel use of the hazard index approach in the proposal is not clearly permissible under the SDWA. The SDWA is designed for an individual assessment of contaminants and an individualized assessment of appropriate MCLG and MCL, as the statute uses the singular "contaminant" when defining "maximum contaminant level."[17] The proposal runs counter to this statutory focus by a Hazard Index approach rather than a specific concentration level for proposing an MCL and MCLG for PFNA, PFHxS, PFBS, and HFPO-DA.

The Hazard Index approach is also arguably inconsistent with SDWA because it is not a "level" but instead a calculated sum of component hazard quotients using a highly variable equation that can change over time.

Earlier in these comments, AWWA raised issues regarding the EPA's occurrence analysis and the proposal's lacking evidence for occurrence, let alone co-occurrence, of these four PFAS. Notably, EPA has not demonstrated that there is or "there is a substantial likelihood that the contaminant will occur in public water systems with a frequency and at levels of public health concern"[18] as required under the SDWA, and at most has shown that they may potentially occur at levels of concern. Additionally, the **agency's approach is contrary to the agency's own guidance for assessing risks of mixtures, which states that it must be "sufficiently similar mixture" where "components and respective portions exist in approximately the same pattern" (EPA, 1986).** As indicated in this guidance, a key feature of a mixture is the mixtures composition and consistent co-occurrence of the components (PFNA, PFHxS, PFBS, and HFPO-DA). The EPA's occurrence analysis fails to sufficiently document co-occurrence of this mixture of PFAS and AWWA's analysis of data from nearly 8,000 water systems does not demonstrate a pattern of co-occurrence.

Additionally, the agency has failed to provide adequate information to support the proposed approach to apply the hazard index to these compounds using reference doses based on the compound-specific critical health outcome, not a similar health outcome. In review of the draft approach, the SAB provided EPA with support for the determination of "dose additivity *based on a common outcome*" while also noting that this was appropriate "instead of a common mode of action as a health protective default assumption" (SAB, 2022). Additionally, the support document for this MCLG states that "component-

[17] 42 U.S.C. § 300f(3).
[18] 42 U.S.C. § 300g-1(b).

based approaches for assessing risks of PFAS mixtures are focused on evaluation of similarity of toxicological endpoint/effect rather than similarity in MOA [mode of action]". The proposed approach is inconsistent with this statement and is contrary to the SAB recommendations.

In addition, ATSDR and EPA guidance on risk assessment for mixtures recommends against the use of a common mode of action when using a hazard index. ATSDR's Framework for Assessing Health Impacts of Multiple Chemicals and Other Stressors notes that the hazard index method is most appropriately "applied to components that cause the same effect by the same mechanism or mode of action" but "may be applied to components with different target organs as a screening measure" (ATSDR, 2018). ATSDR's Public Health Assessment Guidance Manual indicates that when "the health guideline for each contaminant is based on different target organs, health assessors will need to calculate a target-organ-specific HQ [hazard quotient] for each contaminant" (ATSDR, 2022). Both ATSDR guidance and EPA's own guidance for risk assessment of mixtures clearly indicates that the use of critical effects from multiple tissues in a hazard index is generally inappropriate, specifically noting that the index requires "similarity in target organ" (EPA, 1986). In fact, the guidance states that "because the hazard index is tied to a specific effect, the underlying data should be on that effect." As part of this guidance, the agency recommends that target organ toxicity doses as opposed to a critical effect dose, which is what is proposed in this action.

Detailed comments and recommendations on the proposed hazard index approach for PFNA, PFHxS, HFPO-DA, and PFBS were prepared for AWWA by Ramboll US Consulting and can be found in Appendix A of these comments.

AWWA recommends that the EPA re-issue the preliminary determination for PFHxS, PFNA, HFPO-DA, and PFBS as a mixture and recommends that prior to re-issuing a preliminary determination for a mixture of PFAS work towards refining the proposed approach to align with guidance from federal agencies (including the agency itself) and recommendations from the SAB. Aligning the use of the hazard index with agency guidance and SAB recommendations is not only necessary for sound policy, but necessary to comply with both the SDWA and the APA. Further, EPA should follow its own guidance before finalizing any risk assessment for mixtures of PFAS, by EPA ensuring that appropriate peer review be conducted to confirm the agency is relying on the best available science.[19] AWWA notes that EPA followed a more robust process in evaluating PFOA and PFOS and recommends that EPA apply at least the same level of rigor to its analysis of PFHxS, PFNA, HFPO-DA, and PFBS. Failing to do so, or failing to acknowledge this change, and providing a reasoned explanation for the change, would violate the APA.

EPA stated that SDWA provides the agency with the authority to concurrently propose a preliminary regulatory determination and to propose a drinking water regulation for these PFAS, but this is not the case. SDWA provides EPA with the authority to "publish such proposed regulation concurrent with the *determination to regulate*". A determination to regulate is distinctly different from a preliminary determination. Specifically, a preliminary determination applies to the proposed action while a determination to regulate applies to a final action. While this language authorizes EPA to propose regulation as part of the same action as a determination to regulate, it does not authorize that the proposed regulation be concurrent with a preliminary determination.

[19] U.S. EPA, Peer Review Handbook, 4th edition, 2015, available at: https://www.epa.gov/sites/default/files/2015-10/documents/epa_peer_review_handbook_4th_edition_october_2015.pdf.

EPA itself has distinguished between a preliminary and final regulatory determination, the latter of which it interprets to mean the "determination to regulate." [20]

Second, SDWA Section 1412(b)(1)(B)(ii) specifically uses different terms for a determination to regulate and a preliminary determination:

> "Not later than 5 years after August 6, 1996, and every 5 years thereafter, the Administrator shall, after notice of the *preliminary determination* and opportunity for public comment . . . make *determinations* of whether or not to regulate such contaminants." [21]

In addition, collapsing these steps into a single proposal undermines the SDWA's mandate that EPA use the best available public health information to make regulatory determinations in accordance with the three statutory criteria. While EPA is collecting public comment on the preliminary determination (and is seeking more studies and health information from the public), how would it know that regulation is warranted when it lacks a complete record? Instead, the notice and comment provisions exist to allow EPA to collect the data it needs to decide whether to regulate and to ensure the statutory criteria for it to do so are present. EPA's approach is also inconsistent with SDWA Section 1412(b)(4)(C), which states that "[a]t the time the Administrator proposes a national primary drinking water regulation under this paragraph, the Administrator shall publish a determination as to whether the benefits of the maximum contaminant level justify, or do not justify, the costs…" [22] EPA cannot reach such a decision while it is collecting public health data from the preliminary determination phase because it cannot simultaneously determine whether the benefits of regulation justify the costs. And notably, Congress knows how to direct simultaneous regulatory actions under the SDWA when it intends to, but did not do so here. [23]

Because the SDWA does not provide EPA with the authority to propose a preliminary determination to regulate at the same time as a proposed regulation (and as a result it has not been agency practice to do so previously), EPA must re-issue a proposed regulation for PFHxS, PFNA, HFPO-DA, and PFBS after accepting public comment on its preliminary determination to regulate those substances in order to comply with its obligations under the SDWA and APA.

---

[20] *See* 76 Fed. Reg. 7762, 7763 (Feb. 11, 2011) ("What is EPA's final regulatory determination on perchlorate and what happens next?" . . . . "the Agency has made a determination to regulate perchlorate in drinking water [and] EPA is initiating the development of a proposed NPDWR for perchlorate.").

[21] 42 U.S.C. § 300g–1(b)(1)(B)(ii). *See also NRDC v. EPA*, No. 20-133, slip op. at 4 (D.C. Cir. May 9, 2023) ("After the comment period ends, EPA must make its final regulatory determination.") (emphasis added); id. at 11 ("[T]he preliminary determination precedes the notice and comment period. Once that period ends, the agency makes its regulatory determination, and that determination is final.").

[22] 42 U.S.C. § 300g–1(a)(3).

[23] For example, Section 1412(a)(3)) states: "Whenever a national primary drinking water regulation is proposed under subsection (b) for any contaminant, the maximum contaminant level goal for such contaminant shall be proposed simultaneously. Whenever a national primary drinking water regulation is promulgated under subsection (b) for any contaminant, the maximum contaminant level goal for such contaminant shall be published immediately."

7. Compliance Cost Analysis

According to the proposal, EPA anticipates that nearly 67,000 water systems will need to comply with the proposed rule. These water systems will need to review and understand how to implement the rule, conduct initial monitoring of the regulated PFAS at each entry point to the distribution system, and potentially work to install drinking water treatment systems or take another mitigation strategy. The proposal does not fully capture these costs. Because the SDWA requires EPA to take costs into consideration, including when setting the appropriate MCL, and to issue a determination as to whether the benefits of the maximum contaminant level justify, or do not justify, the costs, EPA can only comply with its statutory requirements by conducting an analysis that fully captures these costs and making them available for public comment.[24] The following sections provide recommendations to improve the cost analysis.

Monitoring Requirements for Systems Participating in UCMR 5

Under the proposed rule, community water systems and non-transient non-community water systems will need to conduct initial monitoring, unless the state primacy agency approves the use of previously collected monitoring data. AWWA appreciates the agency's interest in providing this flexibility for water systems that may have already collected PFAS monitoring data for UCMR 5 or state monitoring programs. As part of the compliance cost analysis for the proposal, EPA did not include the monitoring costs associated with certain systems that *may be eligible* to take advantage of this flexibility. While it is reasonable to assume that some systems may potentially avoid initial monitoring for PFAS, it is not appropriate to exclude these costs from the analysis. While all systems serving more than 3,300 are currently already required to monitor for PFAS in accordance with UCMR 5, many of these systems may need to conduct additional monitoring to comply with the rule and to meet the timeline for compliance set by EPA. Examples of these systems may include:

- Large groundwater systems (serving more than 10,000 persons): Large groundwater systems will be required to collect quarterly samples while UCMR 5 requires the collection of two samples for all groundwater systems. There are more than 1,650 systems in this category that would need to collect additional samples beyond UCMR 5 to take advantage of this flexibility. Additionally, systems that are actively collecting samples during 2023 are unlikely to have the opportunity to adjust monitoring plans.

- Water systems scheduled for UCMR 5 monitoring in 2025: The proposal provides a 3-year timeline for water systems to comply with both the initial monitoring requirements and compliance with the MCL. Initial monitoring for the rule will need to begin immediately following promulgation of the rule to ensure that there is adequate time to take necessary action if PFAS levels exceed one or more of the MCLs. Given that the EPA's target date for a final rule is December 2023, all water systems without pre-existing data sufficient to meet these requirements will need to be monitored during 2024. This precludes the use of samples from 2025 under UCMR 5 program, as these results would give these systems less than a year to comply with the three-window for compliance, if treatment was needed.

EPA's WBS Model

[24] *See* 42 U.S.C. §§§ 300g–1 (b)(3)-(4).

According to the proposal, EPA considered three treatment options that may be used by as many as 4,300 water systems to reduce PFAS levels to below the MCLs. These treatment options included GAC, IX, and RO filtration facilities. Additionally, EPA considered other options to address PFAS levels in drinking water, such as interconnections, new wells, and point-of-use RO systems. To estimate the costs to install and operate these systems, EPA relied on their Work Breakdown Structure (WBS) for these strategies, which were developed more than two decades ago and were updated as part of the proposal.

As highlighted by the EPA's "Technologies and Costs for Removing Per- and Polyfluoroalkyl Substances (PFAS) from Drinking Water" document, there are full-scale facilities that are currently using these treatment technologies for PFAS removal. To support EPA's cost analysis, AWWA contracted with Black & Veatch to prepare a cost model (the BV Model) for PFAS treatment using GAC, IX, or RO using their national drinking water treatment expertise and with support from water systems and experts from across the sector (See full report in Appendix B). AWWA has also worked with water systems to compile information on the costs to install PFAS treatment systems.

The BV Model and the case studies were used to compare with the model outputs from the EPA's WBS Model for GAC and IX to better understand the accuracy of the EPA's WBS unit cost models. Figure 7-1 shows an example of a comparison of the available case study data to the BV and EPA cost models, specifically showing capital costs associated with installing GAC treatment facilities for PFAS treatment for systems up to 2.5 MGD.

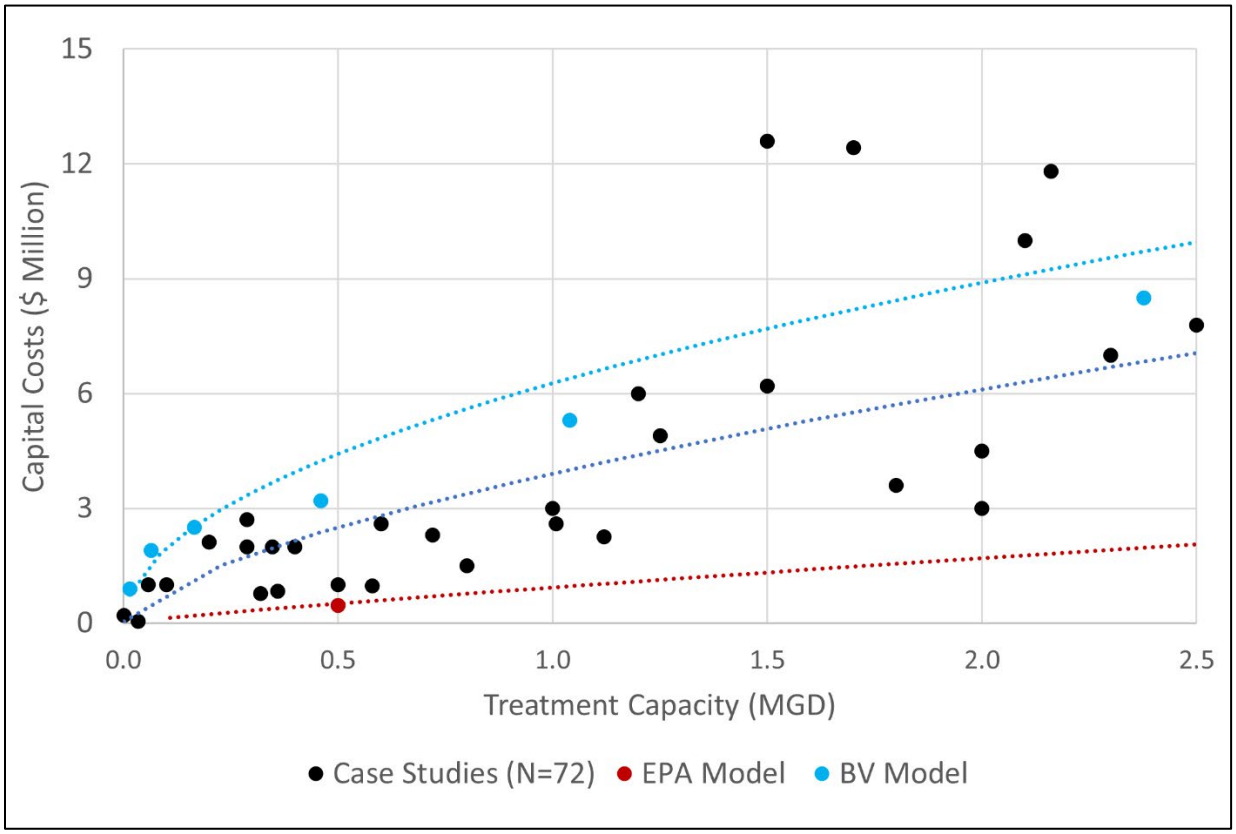


Figure 7-1: Capital Expenses for PFAS Treatment Facilities using GAC Compared to EPA and BV Models

This range was selected as it represents more than 75% of the systems that would be impacted by the proposed rule. Appendix C provides supplemental figures showing a similar comparison for additional ranges of treatment capacity, types of treatment, and operating costs. This data was developed through the use of available data from the Black & Veatch cost model, the example model outputs provided by EPA in the supporting documentation, and more than 100 case studies that were collected earlier this year. For a full list of treatment case studies, refer to Appendix D.

As shown in this figure, the EPA's WBS model significantly underestimates the costs associated with PFAS treatment using a GAC treatment facility. Data from the case studies in this range shows that the typical PFAS treatment system costs 330% more than the estimated cost by the EPA's WBS Model (based on 2021$). The extent of the EPA's WBS Model's underestimation of cost is similarly demonstrated by the BV Model, which shows cost figures in 2022 dollars. These patterns are similarly observed when looking at larger treatment facilities, operating costs, and other treatment technologies. Refer to Appendix B and C for more information.

AWWA also compared the capital cost estimates for GAC under the proposed PFAS rule with the agency's estimates for the Stage 2 Disinfectants and Disinfection Byproducts Rule (EPA, 2005). Under the Stage 2 rule, EPA also estimated costs for systems installing GAC with a 20-minute empty bed contact time. The results of this comparison, Figure 7-1, show an alarming issue: EPA's cost estimate for PFAS removal in 2021 dollars is nearly the same as TOC removal in 2003 dollars. As a point of comparison, the Engineering News Record building cost index has increased from 6,654 in 2003 to 13,288 in 2023 representing a nearly double increase in construction costs alone, which do not include the additional cost increases water systems have been faced with (ENR, 2023).

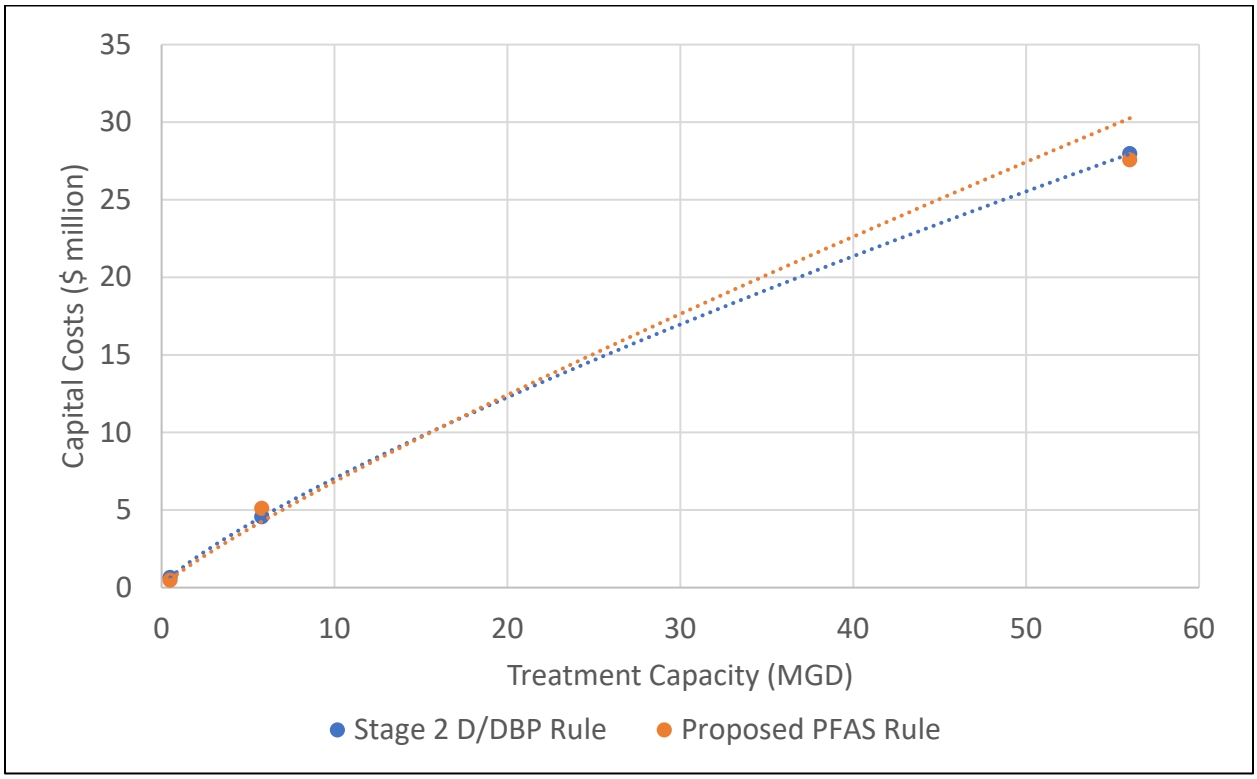


Figure 7-2: Comparison of GAC Capital Costs for GAC for Stage 2 D/DBP and Proposed PFAS Rule

AWWA requested an extension of the public comment period to accommodate a more detailed analysis of the EPA's updated WBS. Given that the agency did not extend the comment period, an exhaustive review of the WBS could not be provided within the 60-day period for review and comment. However, in review of the data from Figure 7-1, Figure 7-2, and the supplemental figures, there is no question that the WBS struggles to accurately capture costs of new treatment facilities. There are some aspects of the WBS that are anticipated to cause a significant underestimation of the true costs of installing PFAS treatment systems. These are further described in the following sections. Given the central role that costs play in EPA's determination, and in order to provide an opportunity for meaningful public comment on the costs associated with this proposal, EPA should work with AWWA and other drinking water treatment experts to revise the cost analysis and provide for an additional comment period on the updated analysis.

General Comments on EPA's WBS Model

Model Accuracy & Contingency

To estimate the costs associated with different treatment strategies, EPA's WBS model uses inputs to prepare a cost estimation for an individual water system. While the model is detailed with respect to potential costs that are considered, there are limitations of this approach, which may require the application of correction factors (similar to the toxicological uncertainty factors).

For each system, EPA estimates the service population using data available through the Safe Drinking Water Information System. The service population provides a reasonable estimate of the general water supply design requirements, which with some broad assumptions (e.g., daily per capita water use, peaking factor, etc) can inform an estimate of the overall water supply capacity needed for each entry point to the distribution system. Then, EPA relies on probabilistic distributions of PFAS and TOC to determine specific unit operation costs for each system in a Monte Carlo Simulation.

Ultimately, the WBS model relies on three key system characteristics to drive the cost estimate: (i) water treatment flow capacity, (ii) PFAS levels, and (iii) TOC levels. This information is used by EPA to estimate the implementation costs for each water system, which is intended to cover planning, design, testing, permitting, and construction of the system. While it is not uncommon for budgetary estimates to be prepared using a limited set of data, it is extremely rare for budgetary estimates of these systems to not recognize that the design is not exhaustive. Good engineering practice is to include an adequate contingency and to transparently describe the level of conservatism in the estimate based on uncertainty in the information available to prepare the cost estimate.

The American Association of Cost Engineering describes five classes of cost estimates that are distinguished by maturity level of project definition, end usage, methodology, and the expected accuracy range. A Class 1 estimate represents a level of project definition exceeding 50% where a detailed unit cost and detailed take-off have been used to estimate the costs and the cost could be as much as 15% higher. Alternatively, a Class 5 estimate represents a level of project definition of less than 2% where concepts are being screened and the use of parametric models were used and so the costs could be as much as 100% higher. In consideration of the data that are available for EPA to consider in estimating costs and given that site-specific conditions cannot be factored into the estimate, the EPA WBS Model is likely to be considered a Class 4 or Class 5 estimate where costs could be 50% or 100% higher.

EPA provides various example model outputs of their cost estimate for different systems, including the estimate of contingency. Table 7-1 provides an overview of the contingency for each of these estimates. As shown in the table, the vast majority of the model outputs show cost estimates for systems with 0% contingency included. This is for 100% of the systems with a treatment capacity below 5.809 MGD, which is pertinent to more than 75% of the water systems that EPA anticipates will be impacted by this rule. For the remaining example outputs, a very low level of contingency is included. This is inconsistent with recommended best practices for cost estimators and is expected to be a major contributor to the EPA WBS' failure to accurately capture costs for PFAS treatment facility implementation.

Table 7-1: Assumed Contingency in EPA WBS Model Example Outputs

Example Output	WBS Estimated System Cost ($)	WBS Contingency ($ / %)
0.500 to 5.809 MGD GAC Systems	$470,000 - $5,117,960	$0 / 0%
56.271 MGD GAC System	$27,557,434	$1,085,355/ 3.9%
0.500 to 5.809 MGD IX Systems	$367,110 - $3,920,210	$0 / 0%
56.271 MGD IX System	$39,765,958	$2,303,916 / 5.8%
0.500 – 5.809 MGD RO Systems	$1,804,014 -$7,613,306	$0 / 0%
56.271 MGD RO System	$36,366,653	$0 / 0%
0.500 – 1.000 MGD Interconnections	$463,316 - $479,635	$0 / 0%
3.536 MGD Interconnection	$2,006,014	$56,535/ 2.8%
0.500 – 1.000 MGD New Wells	$266,255 -$959,977	$0 / 0%
3.536 MGD New Well	$3,040,861	$85,650 / 2.8%

Therefore, the minimal level of contingency, or lack thereof, in the WBS cost estimates wildly overestimates the WBS' ability to capture system-specific water quality, site conditions, community needs, and the overall cost factors for the new treatment facility. The EPA should adjust this approach and ensure that the appropriate levels of contingency are included to ensure that cost estimates are consistent with the level of project definition afforded by the available data in keeping with sound engineering practice.

Accurately Reflecting Current Economic Conditions

Another limitation of the EPA's WBS Model is that it reflects outdated 2021 construction costs. Additionally, the model relies on a variety of cost indices to scale the costs from a previous year to the relevant year of the analysis. It is reasonable to scale data from one year to another year using indices, but EPA fails to account for the fact that there is always a lag in the data for the most recent periods of time. The agency must recognize that the past two years, from 2021, have shown significant cost increases relevant to drinking water treatment systems.

These increases in costs have stemmed from the COVID-19 pandemic, high inflation, and increasing interest rates for borrowing. Construction costs, for example, have increased steadily by more than 15% to 30% in the past 2 years according to several different sources (Mortenson, 2023; Turner Constructon, 2023; USBR, 2023). By comparison, construction costs during 2020 only increased by less than 1.5%. A similar trend can be observed in analysis of inflation since 2021, which has averaged 5.81% annually. The federal funds rate has also increased from 0.08% to 5.25%, which will also drive up the costs for water

systems to secure financing for new projects. EPA's analysis must reflect these increases to properly account for the costs of implementation.

In addition to the increase in costs driven by economic conditions since 2021, it is also important to note that the proposal itself will further increase the costs; 67,000 systems conducting monitoring and upwards of 4,300 water systems installing treatment facility will increase demand for laboratories, engineering consultants, planners, contractors performing site investigation and construction work, and skilled treatment operators.

In order to comply with its statutory obligations, including under the SDWA[25], EPA should ensure that the cost analysis of any rule is accurately reflecting costs due to economic conditions and anticipated increases in demand that will drive the planning and construction costs of new facilities significantly higher than the current estimates.

Recognizing the Importance of Ancillary Systems

Another potential limitation of the EPA's WBS Model is that it only considers capital upgrades for PFAS removal from water. This is a significant analytical gap because many systems will likely need to make improvements to other areas of the treatment facility to support the PFAS treatment process. For example, some systems installing GAC treatment may determine that the concentration and form of manganese will cause problems in the vessel, requiring pre-treatment. A variety of other water quality characteristics may impact the need for pre-treatment and site-specific conditions may drive the need for significant upgrades to critical treatment support systems (e.g., pump stations, chemical feed systems, etc.). It is not uncommon for upgrades for PFAS treatment to require these types of improvements, none of which EPA's WBS Model takes into account.

Lifespan of Treatment Equipment

The proposal provides inconstant information related to the total number of years that are used for the annualization of costs. In the Economic Analysis, EPA notes that both costs and benefits are annualized over 82 years. Alternatively, the example outputs for each treatment system in the Technologies and Costs document lists useful life for each piece of equipment that is included in the capital costs. In the same document a system-specific useful life is listed as part of the cost equations. The supporting documentation from the EPA does not provide a clear explanation of how costs are annualized. Additionally, the useful life varies for equipment from as low as 7 years to as long as 35 years; it is unclear from the supporting documentation the methodology EPA used to substantiate these assumptions. This approach is not consistent with previous practice; the agency's approach to annualizing costs under the Arsenic Rule was based on a 20-year useful life for equipment (EPA, 20XX). In order to fulfill its obligations under the APA, AWWA recommends that EPA provide this information during a supplemental comment period prior to finalizing any PFAS rule. Failing to do so or failing to acknowledge this change in the assumption for treatment facility lifespan and providing a reasoned explanation for the change, would violate the APA.

[25] 42 U.S.C. 300g-1(b)(B)(4)(D).

Interconnections

EPA anticipates that some water systems may take non-treatment actions to respond to PFAS levels above the MCLs, such as the installation of an interconnection. Overall, AWWA agrees that some systems could potentially decide to install an interconnection if it is a viable alternative. Regionalization may have benefits for consecutive systems and can help provide smaller systems with access to economies of scale. Alternatively, regionalization can have unintended consequences on the water quality for the consecutive system, such as elevated water age, nitrification, and DBPs. For this reason, 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.

For the analysis, EPA estimates that upwards of 7% of small systems will install an interconnection to comply with the PFAS MCLs, it is unclear from the supporting information how this assumption was made, and EPA should provide additional information. The potential use of an interconnection to comply with the proposed rule has not previously been included as part of a drinking water rule's compliance analysis and the EPA's approach poses significant issues that exclude significant cost factors. AWWA is providing the following recommendations for necessary considerations for installing interconnections.

- *Compatibility of Secondary Disinfectants*: The use of disinfectants for maintaining a residual in the distribution system varies by system and not all systems use the same disinfectant chemical to maintain distribution system residuals (if they are required to). The selection of a disinfectant for maintaining a residual is not regionally uniform and varies on a number of other factors (e.g., water source, finished water quality, distribution system size, water supply capacity, etc.). Subsequently, there is a significant likelihood that a purchasing system may be using a different disinfectant than a supplying water system, which would require the installation of a facility that can convert free chlorine to chloramine, or vice versa. Some systems may need pH adjustment as well. EPA's cost analysis for interconnections does not consider this. If needed, disinfectant conversion facilities require substantially more upgrades than the EPA's WBS for interconnection considers. Costs for these facilities will need to include, at a minimum:
    - Land purchasing,
    - Building construction,
    - Chemical feed pumps, storage tanks, and spill containment,
    - Mixing and storage tanks, and
    - Water quality monitoring devices.
- *Simultaneous Compliance with LCRR*: LCRR became effective in 2021 and water system compliance with all provisions of the current LCRR will be required in October 2024. Water systems are required to fully evaluate and ensure adequate corrosion control when adding or changing sources of water. Specifically, LCRR expands on existing requirements to include this assessment when adding new sources as previously described. The challenges posed by LCRR will impact the number of systems for which purchased water from wholesale supplier is a viable near-term option. Furthermore, if both systems are using CCT, the compatibility of each CCT must be considered.

- *Simultaneous Compliance with Microbial and Disinfection Byproduct Rules:* As noted previously, regionalization can provide benefits but can also have negative impacts on water quality particularly because of increased water age. For systems that install an interconnection to a consecutive system, a thorough investigation will be needed to determine if water age will be an issue and whether DBPs may need to be addressed at the to prevent MCL violations. This could potentially require systems installing interconnections to install GAC filters and/or transition to a different disinfectant residual. Similarly, systems may determine that inadequate disinfectant residual is present in the water to support the longer water age and so a boosting station is required.

- *Pressure Differences*: In Table 5-15 of the Economic Analysis, the EPA notes that booster pumps and/or pressure reducing valves are included as direct capital costs by the WBS cost model. The agency later notes, however, that to generate cost equations for interconnections the agency has assumed a minimal pressure difference between each water system so that neither booster pumps nor pressure reducing valves are needed. AWWA understands that it may be impossible for the agency to surmise the average pressure difference between two water systems, however it is nonetheless 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. For an interconnection of this distance, the pressure loss associated with water flow through an appropriately sized pipe (to maintain water velocity from 5 to 7 feet per second), would be approximately 50 psi. The inclusion of purchasing booster pumps, at minimum, should be included as part of this analysis.

- *Unit Cost of Purchased Water*: According to the proposal, an assumed average cost of purchased water is $3.00 per thousand gallons (2021$) based on wholesale rates that were available online. Currently there are 3,258 water systems in SDWIS categorized as wholesaler systems. These systems range in service population size from 25 to up to 2.5 million persons, which represents a significant range in their economy of scale. The WBS for non-treatment options and the documentation that is provided does not clearly illustrate what data were considered to estimate the national average cost of purchased water and whether those data are nationally representative. It is possible that if the available data may be only from cities with a water supply that is relatively inexpensive to treat and supply to purchasers. Transparency on this data is necessary to ensure that this unit cost is accurate and reflective of the national perspective. EPA should therefore provide the underlying analysis and an explanation for the model provided in order to allow for a meaningful opportunity for public comment.

  To further illustrate this, a report by the Department of Energy from 2017 assessed water rates nationally and estimated that in 2016 the average water rate was $3.38 per thousand gallons (DOE, 2017). More recent data from Circle of Blue similarly analyzed water rates nationally and estimated that average national water rate in 2019 to be $6.22 per thousand gallons (Circle of Blue, 2019). These national data points highlight a stark difference between the EPA's data and highlight that water rates have increased substantially as water supplies have become more severely impacted by drought and water quality challenges. Additionally, it is important to note that both reports do not reflect water system cost increases related to LCRR, the economic effects of the COVID-19 pandemic (e.g., increased price of chemicals, materials, and labor).

These cost increases have been previously described in this letter. EPA should clearly communicate the sources of the wholesale water rate data so that additional supporting data can be provided to improve EPA's analysis.

- *Available Capacity Without Improvements*: Finally, the EPA's approach makes a blanket assumption that these water systems will be able to identify a supplier water system that has existing available capacity to provide finished drinking water with PFAS levels below the MCL without needing to install any treatment. It is highly likely that a supplier will need to install additional drinking water treatment systems to accommodate the purchasing water system's water supply capacity. This is an especially important consideration for regions of the U.S. where drought is creating water supply challenges already as well as areas where source waters are becoming increasingly challenging with respect to accessibility and ease of treatment. On top of this challenge, there is a significant possibility that the PFAS contamination impacting a purchasing water system is also impacting the supplying water system, especially given the low levels of concern identified by the proposal. In this case, the system providing water supply would need to install PFAS treatment capacity for their current water supply capacity in addition to the purchasing water system's demands.

Finally, in review of the EPA's example outputs for interconnections, the projected costs for a 1 MGD interconnection with mid-cost components were estimated to be less than the projected costs for a 0.5 MGD interconnection with low-cost components. It's unclear whether the model requires correction. Nonetheless, EPA is encouraged to review the model and subsequent cost analysis to ensure that this and other potential errors are addressed prior to using this analysis to support any final rule.

Development of New Wells

EPA also estimates that some water systems will develop new wells instead of installing treatment. The development of new wells also relies on assumed conditions that may make the development of new wells to be more cost effective than treatment. One key assumption is that the PFAS contamination impacting the water system's current groundwater source is not impacting another local source where a new well can be constructed. This is a flawed assumption and likely overestimates the number of water systems for which this is a viable option.

This option also appears to be underestimating the costs for performing this task. In review of the example cost model outputs for a new well of 0.5 MGD, several aspects of the cost estimate are significantly low. A recent budgetary estimate for a water system in Pennsylvania for a new well with a capacity of 0.144 MGD is approximately $1.5 million, which does not include planning and design services totaling another $532,000 (Horsham, 2023a).

The referenced budgetary estimate was compared with EPA's model output for the development of a new 0.5 MGD well. Several aspects of the project are substantially low compared with the referenced estimate. Construction management, for example, is estimated to be less than $16,000 by EPA'S WBS whereas the construction management services for this recently developed well will exceed $175,000. The overall cost of this well will exceed EPA's estimate by a factor of 5 for a well that has less than a third of the capacity. Another water system in Washington submitted comments to the EPA similarly illustrating that these costs are underestimated by a factor of 4 (LWD, 2023).

In order to provide an accurate assessment of costs, EPA must therefore re-evaluate the WBS for new wells and address errors in its estimates prior to using its new well costing analysis to support any final rule.

Social Costs of Carbon Dioxide

To comply with this rule, most water systems with PFAS exceeding the MCL(s) will need to install drinking water treatment facilities that rely on either GAC, IX, or RO. In many cases, it is likely that this will create a new hydraulic profile for the water system which requires additional facility pumping and consequently electricity demand. The use of GAC and IX also requires disposal of spent material involving transport of that material via train or truck to an appropriate facility. These activities can be reasonably anticipated to have a significant impact on the carbon footprint of water systems nationally.

EPA is currently heavily involved in addressing challenges with climate change and in advancing sectors of the U.S. economy towards reducing greenhouse gas emissions. Section 5 of President Biden's Executive Order 13990, notes that it is "essential that agencies capture the full costs of greenhouse gas emissions as accurately as possible, including by taking global damages into account" and that doing so "facilitates sound decision-making" (Biden, 2021).

The proposal lacks an analysis of the social costs of carbon. AWWA recommends 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.

In considering the social costs of carbon, the agency is encouraged to review a recent report by Policy Navigation Group (PNG, 2023). The current estimate for the social cost of carbon dioxide emissions varies, but a low estimate range is $130 to $190 per ton based on a recent EPA report (EPA, 2022h). In the Policy Navigation Report, they estimate the social costs of carbon using data from the EPA's current economic analysis and using available EPA guidance for estimating such costs. Policy Navigation Group estimates carbon emissions related to additional pumping, lighting, and ventilation associated with the PFAS proposed rule and concludes that the potential national social costs of the carbon emissions are $5 million annually. The $5 million would be in addition to the social costs associated with replacement of GAC and IX media as breakthrough occurs. Given that more than 4,300 water systems will rely on GAC and IX treatment for PFAS and will begin generating tens of thousands, if not hundreds of thousands of tons of spent GAC and IX resin it is important that the associated social costs are considered.

EPA estimated the total GAC and IX waste generation annually to perform a sensitivity analysis for managing these materials as hazardous wastes, but the estimated amount of waste generated is not reported. EPA estimated that water systems will need to install treatment for more than 64.8 million people to comply with the proposed rule; this will amount to more than 3 trillion gallons that will need to be treated annually. Calgon Carbon estimated one water treatment plant's GAC usage rate for PFAS treatment was as high as 0.07 pounds GAC per 1000 gallons (Calgon, 2023). Based on these figures, the annual demand for GAC could exceed 100,000 tons, which potentially has a carbon dioxide footprint of 850,000 tons (He, 2012). This could have a social cost of more than $160 million annually.

With such a significant potential impact on society, EPA should conduct the same analysis to determine the social costs of carbon associated with each of the treatment technologies and the rule options. This

analysis should also be included as a matter of maintaining consistency across the agency's rulemaking processes. In two recent rulemaking, EPA estimated the social costs of the rule in recognition that changes to the operation of complex treatment systems can provide both benefits and unintended consequences (EPA, 2023f; EPA 2023g).

Shifting Landscape of Residual Management Practices

During the stakeholder engagement in advance of publishing the proposal, EPA was encouraged to consider the impacts of new regulatory actions that would impact disposal of GAC, IX, and RO waste streams with PFAS. In September 2022, EPA proposed to designate PFOA and PFOS as hazardous substances under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA); a broader action was proposed for additional PFAS in April 2023 (EPA, 2022i; EPA, 2023h). Separately, EPA is also preparing to list PFOA, PFOS, PFBS, and HFPO-DA as hazardous constituents (EPA, 2022c).

EPA considers these actions as part of the economic analysis, particularly through a sensitivity analysis. As noted above, the quantity of GAC and IX that EPA estimates will be disposed of annually is not reported in the supporting information. As noted above, the demand for GAC may exceed 100,000 tons annually. EPA has recently estimated that the costs to incinerate hazardous waste ranges from $400 to $1,700 per ton depending on the type of waste (EPA, 2020). At a minimum, the annual costs to incinerate GAC and IX as hazardous waste will exceed $40 million and are likely to be beyond $100 million. Just the costs of incineration significantly exceeds the $30 million estimate, which EPA describes as including hazard waste disposal and associated costs (e.g., transportation and handling).

While EPA notes that hazard waste disposal costs are excluded from the estimate of annual costs, these costs should not be ignored. EPA's current commitment is to finalize the PFOA and PFOS hazardous substance designations under CERCLA this summer. Furthermore, as these actions are advanced, waste management practices are shifting and leading to increased disposal prices for water systems and, in some cases, facilities are refusing to accept water treatment residuals (AWWA, 2022).

AWWA recommends that the EPA (i) more closely look at the costs associated with hazardous waste disposal, including available guidance from Office of Land and Emergency Management (OLEM) and (ii) include the cost of hazard waste disposal of spent media. The current EPA regulatory agenda will drive impacts the cost of treatment options for removing PFAS from drinking water and will be legally binding by the time water systems must comply with any PFAS rule.

Importance of an Accurate Cost Analysis

The EPA is encouraged to review the cost analysis, and the EPA's WBS Model, to ensure that the monitoring and treatment costs are accurate. As noted above, EPA cannot fulfill its obligations under the SDWA unless the cost assessment is accurate. In addition, this cost analysis is critical given that it underpins the HRRCA as well as the household affordability analysis. An inaccurate estimation of costs will mischaracterize the proposal's regulatory impact, merits, and the affordability for households in smaller communities. These analyses are critical for the agency, and water systems, to understand public health priorities. Water systems are currently working to address various public health priorities, including lead service lines, cybersecurity, microbials and DBPs, water supply challenges (such as drought, increasingly impaired water sources), and aging infrastructure.

Black & Veatch estimates that the cost of rule will range from more than $2.5 to $3.2 billion annually (Black & Veatch, 2023 – See Appendix B). As proposed, this rule will be one of the most costly rules under SDWA ever proposed and will be a burden carried by less than 10% of water systems. It is imperative that the overall analysis of the proposal's impacts be accurate so that water systems and communities are assured that the investments in infrastructure being made represent the investments with the greatest societal benefit.

8. Health Risk Reduction Analysis

As required by SDWA, EPA has prepared a health risk reduction analysis for each of the proposal's regulatory options. AWWA contracted with Ramboll Consulting U.S. to assist in reviewing the EPA's approach and to offer detailed recommendations to improve this important work. Ramboll has provided a detailed letter with recommendations, which is included in these comments in Appendix A.

AWWA requests that EPA update its health risk reduction analysis in light of Ramboll's report and the recommendations it contains so as to ensure that the agency is relying on the best available public health information in reaching any regulatory decisions. MCLs are appropriately set at a level where the benefits justify the costs, and without a reliable assessment of both the costs and benefits of the proposal, EPA cannot do so. Based on the information provided in the proposal, the benefits do not justify the costs at the proposed MCL levels and particularly do not justify the costs under EPA's proposed compliance timeline.

Estimating Reductions in Cardiovascular Disease Risks

According to the proposal, increased PFOA and PFOS serum concentrations may lead to an increased risk of cardiovascular disease (CVD), including myocardial infarctions and strokes. To support this analysis EPA relies on an assumption that there is a causal relationship between PFAS exposure and CVD. In particular, this analysis uses exposure-response functions from a meta-analysis that was conducted using data from several epidemiological studies on the general population. In review of this approach, several critical issues have been raised:

- While EPA relies on epidemiological studies of participants from the National Health and Nutrition Examination Surveys (NHANES) to support its meta-analysis of PFAS exposure and CVD risk. These studies did not observe increased risks of CVD in the participants of these studies, including those participants with highest exposures yet EPA characterizes the results of these studies as 'inconsistent'.

- The analysis of CVD risk projects changes to total cholesterol and blood pressure based on PFOA and PFOS exposure but excludes changes to high density lipid cholesterol, which has also been observed. A sensitivity analysis of a hypothetical exposure reduction of 1 ppt PFOA and PFOS, which included the effects of changes to HDL-C, found that annualized CVD risk reduction benefits were decreased by 23 to 25%. By comparison, when changes to blood pressure were excluded the reduction in annualized CVD risk reduction benefits only decreased by 1.8 to 2.3%. This suggests that the benefits analysis, which excludes the impact of PFOA and PFOS on HDL-C, is significantly overestimated.

- Throughout the supporting documentation, the EPA makes contradictory statements about the strength of associations of PFOA and PFOS on blood pressure and HDL-C. Furthermore, the

inclusion of blood pressure impacts from PFOA and PFOS exposure but not the inclusion of changes to HDL-C is unclear, especially given that these impacts were observed equivocally.

- The biological mechanism for the association of PFOA and PFOS with cholesterol is not yet identified in humans. In fact, recent work demonstrates that a lifestyle intervention on cholesterol led to a decrease in both the cholesterol and serum PFOA and PFOS concentrations.

- Finally, given the recent 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 is reported as part of this proposal, given the outsized impact from other risk factors.

Additional details on these and other comments for the EPA's analysis of cardiovascular disease risk reduction are available in Appendix A. Based on the materials made available, it does not appear that EPA's conclusion is fully supported by the evidence or record before the agency.

Estimating the Reduced Impact of Low Birth Weights

In review of the EPA's approach to estimating the benefits of reducing the incidence of low birth weights resulting from prenatal PFOA and PFOS exposure, it appears that EPA is conflating risk of low birth weight with differences in mean birth weight. In review of this approach, several critical issues have been raised:

- The use of low birth weight as the critical effect is inconsistent with other regulatory agencies that found small decreases in birth weight but not increased risk of low birth weight in relation to PFOA and PFOS (ATSDR, 2021; EFSA, 2020; EFSA, 2018).

- In deriving the Reference Dose for both PFOA and PFOS, EPA has noted that the derivations for PFOA and PFOS were based on low birth weight (defined as birth weight less than 2,500 grams). However, the exposure-response coefficients used for these efforts were based on decreases in birth weight. In particular, the studies used for these derivations evaluated differences in average birth weight but not risk of low birth weight. EPA's conflation of low birth weight with decreases in birth weight is prevalent in the economic analysis as well. While these endpoints are correlated, they are not equivalent and the should not be evaluated as if they are the same.

- The benefits analysis relies on exposure-response functions based on coefficients for decreases in birth weight from the main analysis of two different meta-analyses for PFOA and PFOS. The critical study that serves as the basis for the PFOA meta-analysis (Steenland et al, 2018) concluded that there was no effect on birth weight when the results of the C8 Science Study were included and noted that the results were consistent with confounding and/or reverse causality. EPA's conclusion is at odds with Steenland et al's own conclusion for their work.

- EPA's characterization of the supporting studies for this health effect is inconsistent with the data that is provided by the studies. In the toxicity assessment for PFOA, EPA concludes that the majority of the studies considered showed supportive evidence of an increased risk of low birth weight with increasing PFOA exposures. However, closer inspection of the studies showed conflicting results. For example, several studies that stratified results by sex provided mixed

results on gender-specific impacts. Similarly mixed results were noted for studies that stratified results by country of birth. Furthermore, exposure-response results were not consistent with results from studies where exposures were measured in the general population; in particular, these studies generally reported no associations or very limited evidence of associations of PFOA and PFOS with low birth weight.

- While the SAB requested that USEPA reevaluate and consider studies published before the 2016 Health Effects Support Documents for PFOA and PFOS, at least seven studies that were available prior to 2017 were not included (Darrow et al. 2013; Nolan et al. 2009; Savitz et al. 2012a, 2012b; Stein et al. 2009; Chen et al. 2012; Wu et al. 2012). None of these studies found an increased risk of low birth weight in relation to PFOA or PFOS. EPA must include studies that do not support its preferred outcome in its analysis and explain why, in light of these studies, it has still reached its determination.

- The EPA also failed to acknowledge the significance of the results from the sensitivity analysis assessing bias associated with timing of the maternal sampling during late pregnancy versus sampling during early pregnancy. This sensitivity analysis highlighted that there was no effect on birth weight when maternal blood was sampled early in pregnancy whereas late pregnancy sampling showed a larger effect on birth weight than the meta-analysis. This suggests that the evidence for a conclusion that PFAS exposure is associated with decreased birth weight is inconsistent after considering potential confounding.

Additional details on these and other comments for the EPA's analysis of decreased birth weights are available in Appendix A. Based on the materials made available, it does not appear that EPA's conclusion is supported by the evidence or record before the agency.

9. <u>Monitoring Requirements</u>

The agency is providing a compliance timeline of three years for water systems subject to the rule to perform initial monitoring requirements for PFAS at each entry point to the distribution system. The initial monitoring requirements may be waived for some systems that are either participating in UCMR 5 or have participated in eligible state monitoring programs since January 2019. Initial monitoring will determine if systems are eligible for a reduced monitoring frequency under the proposed framework and if the system will need to install treatment (or take non-treatment action) to reduce PFAS to levels below the MCL. According to the proposal, compliance with the proposed rule will be determined using the RAA at each entry point to the distribution system. As part of the calculation of the RAA, EPA is also proposing that systems use 0 ppt for results that are below the practical quantification limit (PQL) of 4.0 ppt. At the same time, EPA is proposing that to calculate the RAA for determining a system's eligibility for reduced monitoring, that only values below the detection limit be considered as 0 ppt and all reported results above the detection limit be used. The following sections provide a detailed review of these requirements and their proposed alternatives.

Initial Monitoring: Use of Existing Data and Timeline

AWWA appreciates the agency's interest in reducing public water system monitoring burdens, especially where existing monitoring data exists. The agency's proposed approach to accept data collected since January 1, 2019 is appropriate. Reduced regulatory monitoring demands will be a welcome relief for

water systems already working to understand PFAS levels in their water, but it is uncertain what magnitude of impact this will have on water systems.

It is difficult to estimate the magnitude of the reduced monitoring impact because data will be (i) dependent on fully fulfilling the initial monitoring requirements and the quality control that they require and (ii) subject to state primacy agency approval. For example, many states may have data for PFOA, PFOS, PFHxS, and PFNA but not PFBS and HFPO-DA. In other cases, water systems may not have received results for PFAS below the proposed PQL. Another challenge for systems to use previously collected data, such as data from UCMR 5, is that the results may not have been reported at the necessary level to be used for EPA's monitoring and will likely not be feasible for all water systems to acquire data below the PQL.

Additionally, the timeline for the initial monitoring period of three years is appropriate. It is important to note, however, that this timeline is in concert with the compliance deadline for meeting the MCLs. As such, water systems will not be able to wait until the third year of the compliance window to perform initial monitoring requirements because this would not leave time to comply with the MCLs, if elevated levels of PFAS are found.

Use of Standard Monitoring Framework

The proposed approach to use the SMF as a basis for triggering reduced monitoring is appropriate. In review of the proposal, however, AWWA has significant concerns surrounding the proposed approach to require water systems to consider low quality, unreliable analytical results below the PQL. Specifically, the EPA is proposing that 1.3 ppt PFOA and PFOS and one-third of the hazard index be used as a trigger level for reporting for the purposes of determining reduced monitoring eligibility under the SMF. The EPA also requested input on the use of 2.0 and one-half the hazard index.

AWWA recommends against the use of any data below the PQL to drive regulatory requirements. Data should likewise not be used to determine nor treatment, or more frequent monitoring. For the reasons detailed below, use of this below-PWL data would be arbitrary and capricious.

The PQLs for PFOA and PFOS are set at 4.0 ppt each in the proposal, which is consistent with the currently active UCMR 5 monitoring program's minimum reporting limits. In finalizing the UCMR 5 monitoring program in 2021, EPA recognized that while EPA Methods 533 and 537.1 can both be used by laboratories to achieve lower reporting limits but concluded that the available lab capacity could not support establishing lower reporting limits to collect national occurrence data. Data below this level is less accurate and is not achievable by all water systems.

As a matter of policy, EPA should not set a precedent for the use of analytical results that are not reliably achievable for all water systems as this would create an equity issue. Moreover, the current minimum reporting levels for EPA Methods 533 and 537.1 are appropriate based on ongoing experience with PFAS analytical results.

As PFAS occurrence data are required to be collected and used to drive decision-making, there will be higher demand for laboratories that are able to analyze samples with a greater degree of reliability at single digit, part per trillion levels to minimize the risks of inaccurately higher sample results from interferences and other technical challenges. Increased demand for better laboratories will contribute to higher lead times, per sample costs, and more frequent recognition of sample analysis errors. The

consequences of these pressures on the analytical services market, will most negatively impact smaller systems with less financial capacity to access more experienced and better performing laboratories. To the degree small systems have limited access to high quality laboratory service, it creates inequitable access to reliable sample analysis.

Beyond the technical challenges of this aspect of the proposed monitoring requirements, there are challenges with the risk communication of associated results it would produce. As proposed, EPA would require that PFAS monitoring levels be described in two different ways. Risk communication will be especially challenging for water systems with observed values below the PQL, as they will have a reporting PFAS values based on the RAA for MCL compliance and a separate value for reduced monitoring eligibility. It is unclear if EPA has considered how these data would be reported and communicated to the public in a meaningful manner.

Analytical results below the PQL should not be used, rather the rule requirements should use 0 ppt for all analytical results below the PQL. Consistent with previous AWWA comments, AWWA recommends that one-half the MCL be used to determine if PFAS levels at an entry point are reliably below the MCL.

10. Public Notifications

Public notifications serve an important role in protecting public trust in drinking water and their usefulness for communities relies on a solid foundation for risk communication. The proposal requires that water systems with PFAS levels exceeding the MCLs must provide a Tier 2 notification to the public and Tier 3 notifications when a monitoring and/or testing procedure violation has occurred. Water systems will also be required to include information about detections of regulated PFAS in the Consumer Confidence Report.

AWWA supports the proposed approach for public notifications for PFOA and PFOS. While AWWA is recommending in this letter that the proposed regulation for PFHxS, PFNA, HFPO-DA, and PFBS be revised based on the discussed technical and legal issues, the agency is reminded that the risk communication for PFAS must be carefully and thoughtfully structured. The role of public communications is to provide useful information to the public about their drinking water. As EPA considers any rule for PFAS through a hazard index, it is important that regulations be structured in a manner that facilitates useful risk communication. EPA's proposed use of the general hazard index combining risks across multiple health outcomes prevents water systems from having a effective risk communication strategy. If EPA moves forward with any rule using the hazard index, risk communication should be considered more carefully.

11. Household Affordability and Small System Compliance Technologies

As noted in the first section, it is critical that drinking water be affordable and that smaller systems more susceptible to affordability challenges have access to compliance technologies. This is true for all consumers, and particularly for those in environmental justice communities. The proposal highlights several variations of the household affordability analysis beyond the EPA's previously utilized approach, including an approach that has been developed and recommended by AWWA and other water sector associations (AWWA, 2021b). AWWA appreciates that the agency is interested in utilizing recommendations previously made by stakeholders regarding alternative metrics for this analysis.

These comments have already highlighted significant concerns about EPA's underlying approach to the cost analysis and the anticipated inaccuracies of the EPA's WBS Model. EPA should refine that approach and re-evaluate the affordability analysis for any rule.

As part of its analysis, Black & Veatch assessed household impacts of the various rule options (Black & Veatch – See Appendix B). Table 11-1 provides an overview of those results for Options 1a (4 ppt PFOA and 4 ppt PFOS) and 1c (10 ppt PFOA and 10 ppt PFOS) in comparison with EPA's estimated expenditure margins for the affordability analysis. As shown in Table 10-1, the household costs for each of these options significantly exceed the expenditure margin for systems serving less than 1,100 persons.

Table 11-1: Comparison of EPA Affordability Margin and Treatment Cost Estimates

Population Range	Household Costs Estimated by BV Model	EPA Estimated Expenditure Margins
25 to 100	$3,570	$877
101 to 500	$1,675 - $1,750	$877
501 to 1,100	$1,360 - $1,390	$753
1,101 to 3,300	$575 - $640	$753
3,301-10,000	$305 - $327	$855

It is also important to note that these household costs are only reflective of treatment costs, monitoring costs are not included here, which for smaller systems will have a greater household impact. A treatment technology that is not considered as part of this analysis is the use of point-of-use reverse osmosis (POU RO) systems. While POU RO may become available in the future following NSF/ANSI certification standard that is based on achieving levels at or below the proposed MCLs, it is currently not a compliance option. AWWA agrees with the agency's decision to not include POU devices in its analysis of rule compliance affordability for small systems. Certification is currently not available, and demonstration of effectiveness is a critical aspect of including compliance technologies in this analysis. NSF/ANSI certification will be necessary if any rule considers POU RO as a small system compliance technology.

EPA should re-consider the proposal as the small system household costs for centralized water treatment exceed EPA's estimated expenditure margins for these systems, a more affordable POU treatment option is not available, and EPA has not identified a small system variance technology.

Accounting for Financial Assistance

In its affordability analysis, EPA also cites that an additional analysis was conducted accounting for funds that are nationally available, such as the DWSRF program and funds from BIL. As EPA notes in the proposal, $800 million is available annually for systems addressing emerging contaminants like PFAS. EPA also announced the availability of $1 billion annually through the Emerging Contaminants in Small or Disadvantaged Communities grant program. Both programs are appropriated through Fiscal Year 2026. These programs make $1.8 billion available for fiscal years 2024, 2025, and 2026 (a total of $5.4 billion).

The EPA estimates that the total capital cost needs for small systems will range from 1.1 to 2.5 billion; however, as previously noted the cost analysis used by the EPA is significantly flawed and

underestimates financial impacts on communities. The occurrence analysis also presents several issues in characterizing the total small system impacts. For a more accurate comparison to available funding, and therefore potential offsetting of household costs, data from Black & Veatch was considered. Black & Veatch estimates that the total capital cost for small systems for a 4 ppt PFOA and 4 ppt PFOS rule will exceed $21.6 billion based on the occurrence data collected by Corona Environmental (Black & Veatch, 2023; Corona, 2021). This is a stark difference from EPA's estimate of $1.1 to $2.5 billion total capital cost for small systems.

Even if the occurrence analysis from the EPA is used these estimates are substantially low. Policy Navigation Group estimated the number of systems that would potentially exceed the MCLs using data from the EPA (PNG, 2023). Using these figures and the estimated capital costs for each system size from Black & Veatch, the total capital cost exceeds $10 billion. This is approximately more than the EPA's estimate by a factor of four.

The availability of $5.4 billion for these systems will help alleviate the costs for individual households. That impact, however, is limited to systems that receive financial assistance. Unless EPA plans to work with states to develop a method for distributing these funds equally to all impacted water systems, the financial assistance will not be evenly distributed across small systems. This approach will inaccurately depict the financial impacts to households in communities where financial assistance is not provided. Therefore, AWWA recommends that the EPA not consider this financial assistance in assessing household affordability for small systems.

12. <u>Executive Order 12898 – Achieving Environmental Justice</u>

AWWA and our members have first-hand knowledge and experience of how the increased costs associated with new regulations such as the ones proposed here directly impact the customers of our water system members. Many water systems are small, public, or quasi-public entities. Increased compliance costs are necessarily passed on to customers in the form of high rates for their drinking water. As a result, unjustified compliance costs have a disproportionate impact on economically disadvantaged customers as they are least able to afford these rate increases. This is disproportionate impact is particularly acute with respect to water infrastructure for several reasons. First, water systems serve local customers, they do not have the ability to spread costs out across a national customer-base. Second, because household water use is a necessity and most households cannot meaningfully scale back on their needs for drinking water when prices increase. Consequently, they are unable to take steps to reduce their water bills when water systems are forced to increase rates.

Congress amended the SDWA in 1996, recognizing that the Act's prior requirements, and associated economic burdens on water systems and the States were making the SDWA unworkable.[26] As a member of Congress explained at the time, "[c]ustomers will pay for safe drinking water . . . [b]ut are not willing to pay for complying with drinking water rules that provide only marginal increases in health protection

[26] S. Rep. No. 104-169 at 2, 11, 17 (1995); H.R. Rep. No. 104-632 at 9 (1996); *see also* S. Rep. No. 104-169 at 12−13 (noting that the prior version of the Ac was the "quintessential example of an arbitrary Federal law imposing burdens on consumers and the taxpayers of other governments with no rational relationship to the public benefits that might be realized.").

at significant costs, particularly when there is so much uncertainty concerning both the occurrence and real threat to public health of many contaminants." [27]

EPA's current analysis also fails to consider how these increased compliance costs will impact environmental justice communities, as required by Executive Order 12898. As you know, Executive Order 12898 directs each Federal agency to "make achieving environmental justice part of its mission by identifying and addressing, as appropriate, disproportionately high and adverse human health or environmental effects of its programs, policies, and activities on minority populations and low-income populations." EPA should revise its environmental justice analysis to reflect the burdens that the compliance costs associated with the new proposed requirements would place on environmental justice communities and further consider whether these additional burdens are appropriate in light of these impacts.

AWWA further notes that prematurely issuing national primary drinking water regulations for contaminants when the occurrence data indicates that it only occurs in drinking water at levels of public health concern in localized areas would cause communities, rather than those responsible for the pollution, to foot the bill for the problem. To avoid this inequitable result EPA should focus on using its other authorities to address any necessary clean ups, rather than pass the costs on in the form of higher rates due to increased SDWA compliance costs.

13. Alternative Regulatory Options for Drinking Water Standards

The proposal provides a detailed overview of the four regulatory alternatives that were considered by the Administrator to support this rulemaking. These options are laid out both in the preamble of the proposal and the economic analysis. The economic analysis frames both the costs and benefits of the proposed option and its alternatives. The proposed option is 4 ppt PFOA, 4 ppt PFOS, and a hazard index for PFHxS, PFNA, HFPO-DA, and PFBS of 1.0. Option 1a, 1b, and 1c propose MCLs for PFOA and PFOS each at 4 ppt, 5 ppt, and 10 ppt, respectively. Figures 13-1 and 13-2 depict the results of EPA's analysis based on both discount rates of 3% and 7%. Presentation of both discount rates to inform decision-making is consistent with Office of Management and Budget guidance for regulatory analyses.

A striking observation from both figures is the difference in annualized costs for the Proposed Option and Options 1a and 1b compared with Option 1c. Under both discount rates, the cost to implement the rule doubles from Option 1c to 1b. This is reflective of the significant number of systems that would be required to install PFAS treatment facilities to mitigate PFAS to comply with rule Option 1b compared to Option 1c. This difference is expected to impact systems that have PFAS levels ranging from 4 to 8 ppt, as EPA assumes that only systems within 80% of the MCL will make costly infrastructure investments.

These figures demonstrate that, regardless of the discount rate, there are minimal incremental benefits with the addition of the hazard index MCL for PFHxS, PFNA, PFBS, and HFPO-DA. This is substantiated by the available occurrence data, which demonstrates that very few systems would be required to install treatment for the PFHxS, PFNA, PFBS, and HFPO-DA MCL but not the PFOA and PFOS MCL.

[27] H.R. Rep. No. 104-632 at 9 (quoting Ronald Dungan, President of the National Association of Water Companies).

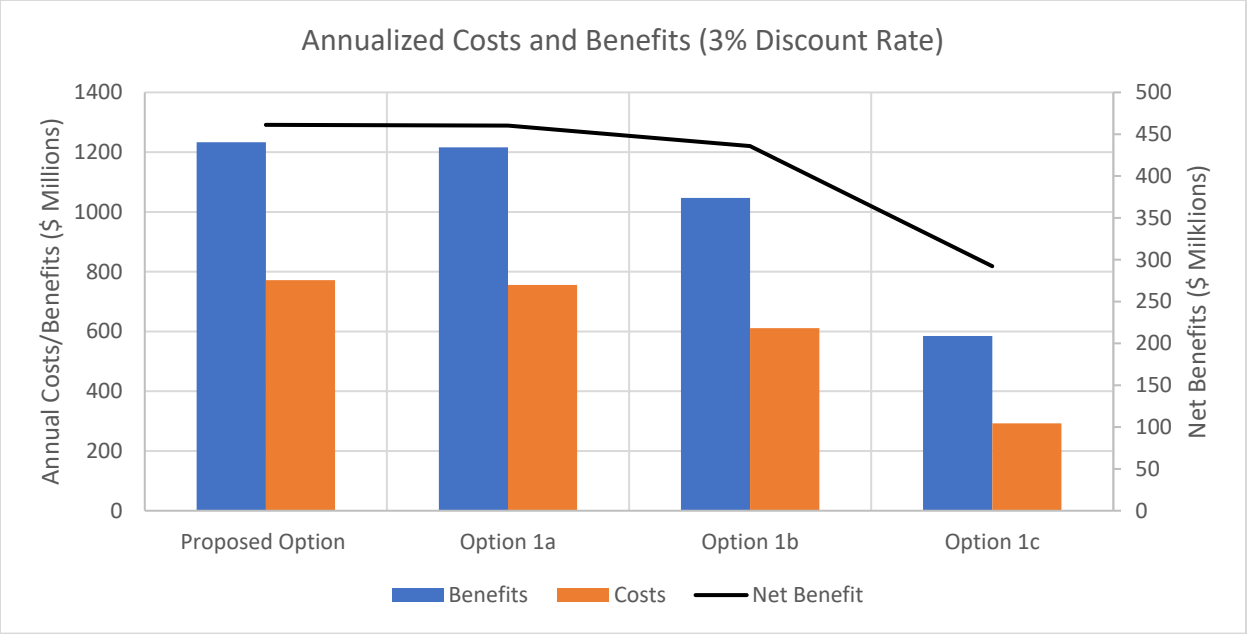


Figure 13-1 : Annualized Costs and Benefits (7% Discount Rate)

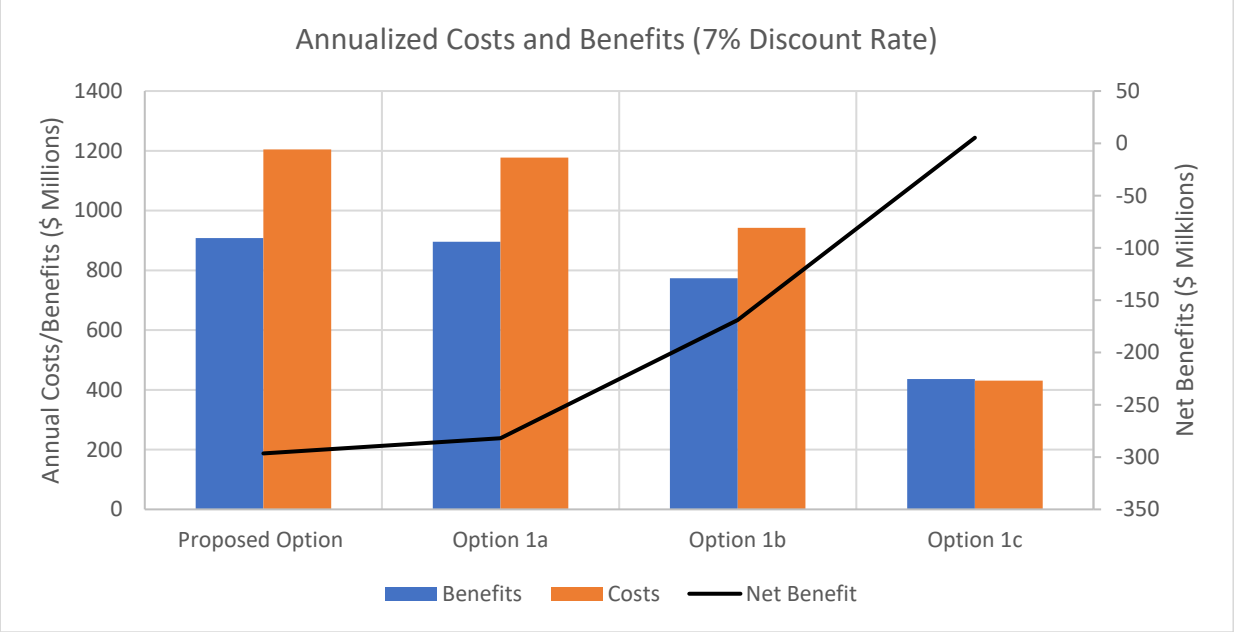


Figure 13-2 : Annualized Costs and Benefits (7% Discount Rate)

To evaluate each regulatory option, AWWA considered the various concerns with the underlying analyses to support the rule that are discussed throughout this letter and the guiding principles for PFAS regulation that were noted earlier, and has determined that any rule will require a substantial level of work to improve the analyses that support the decision by the EPA. Furthermore, AWWA recommends that – if any rule is finalized based on the proposal – EPA finalize the MCLs for PFOA and PFOS at 10 ppt each. AWWA makes this recommendation for several reasons, laid out in more detail below.

The EPA's proposed approach (4 ppt PFOA, 4 ppt PFOS, and hazard index of 1.0 for PFHxS, PFNA, HFPO-DA, and PFBS) will not be feasible, as they will create significant challenges for water systems to implement, is not clearly EPA's legal authority under SDWA, and rely on a series of critically flawed analyses that mischaracterize the impacts of the proposed rule. As discussed earlier, the large number of systems that will need to install drinking water treatment will create challenges in implementing the MCLs on the timeline provided by the EPA and, ultimately, systems will not be able to meet these timelines while also applying best engineering practices to plan, design, and construct these facilities. Additionally, the EPA's statement that it has legal authority to concurrently propose a drinking water regulation with a preliminary determination is a misinterpretation of SDWA. Finally, the hazard index approach is insufficiently substantiated by occurrence data and toxicological science.

As with the EPA selected rule option, setting the nation's first drinking water MCLs for PFOA and PFOS at levels as low as 4 or 5 ppt will create a significant combination of implementation challenges. The effect of such a rule option selection will be to delay all water systems in installing treatment facilities and increase the burden on households to pay for doing so, rendering such levels infeasible. Instead, EPA should place an emphasis on higher-priority water systems by targeting those systems higher levels of PFAS first. Focusing on systems with higher levels of PFAS will ensure that communities with the greatest risk of exposure to PFOA and PFOS are able to control exposure via drinking water more cost-effectively and promptly. Setting the nation's first drinking water standard for PFOA and PFOS at 10 ppt does not preclude EPA from further reducing these standards as technology advances and available occurrence and toxicological data improve.

Ultimately, each of the proposed options is likely to represent a net cost to society (and drinking water consumers), requiring investments that will outweigh the benefits. These impacts will be most dramatically felt by smaller systems serving less than 10,000 people and the affordability analysis suggests that the costs to implement these treatment facilities will range from hundreds to thousands of dollars annually for individual households, significantly exceeding affordable margins for household expenditures for drinking water (i.e. drive the cost of water services beyond EPA's measure of affordable drinking water).

Consequently, EPA should significantly improve upon the analyses to strengthen any rule and to accurately capture the impacts on water systems and public health. The Administrator should re-evaluate this rule with the improved analyses to determine if the benefits justify the costs and that the rule is feasible for small systems to implement. If any rule is finalized without the additional analysis and public review AWWA recommends that EPA utilize Option 1c and set MCLs of 10 ppt PFOA and 10 ppt PFOS. Option 1c affords the greatest opportunity for health benefit for impacted communities while reducing affordability concerns associated with the rule. If EPA determines that regulation of additional PFAS is merited, the agency should propose a rule following a final determination to regulate, consistent with the authority provided by SDWA.

14. Summary of Key Recommendations

Advancing public health is a shared goal between drinking water systems and EPA and AWWA has evaluated the rule to support EPA's actions moving forward meaningfully and legally, using sound science. AWWA appreciates the agency's interest in preparing a thoughtful, thoroughly crafted proposal to establish NPWDRs for PFOA, PFOS, and additional PFAS.

As previously discussed, the regulation of PFAS in drinking water as proposed will create numerous implementation challenges. Although EPA is interested in an expeditious rulemaking to reduce PFAS exposure, it is nonetheless important that EPA finalize a rule that is based on sound science, recognizes the importance of drinking water affordability, and be feasible to implement.

While EPA has a stated interest in advancing immediate protection of communities from PFAS exposure, water systems still need to perform the necessary work to implement any rule requirements – regardless of the timeline. Three years is insufficient time for water systems to comply with EPA's proposed rule option. The Administrator should provide a 2-year extension as part of the final rule per authority provided by SDWA, instead of relying on already overextended state primacy agencies.

Several of the EPA's analyses underlying the rulemaking need improvement to be credible. The occurrence analysis lacks transparency on the levels of PFAS in communities nationally and criteria for data inclusion/exclusion is not clear. It is also an overly complicated approach to assessing national occurrence data for PFAS, which are currently being collected as part of the UCMR 5 program. The cost analysis for drinking water treatment is demonstrably underestimating the impacts of the rule based on case study data and a model by Black & Veatch, which was crafted leveraging long-standing national PFAS treatment design expertise. Finally, the benefits analysis includes several assumptions that are not clearly and consistently discussed by the agency. For example, the analysis of CVD risk reduction accounts for impacts to total cholesterol from PFAS exposure but excludes the impacts to HDL-C, which decreases risks of CVD. These analyses, which underpin the rulemaking's HRRCA and affordability analysis, need significant improvements.

As part of this proposal, EPA proposes preliminary determinations for PFHxS, PFNA, HFPO-DA, and PFBS concurrently with a proposed drinking water standard for these compounds. The preliminary determinations for PFNA, HFPO-DA, and PFBS are not supported by the available occurrence data and the determination for the mixture of these PFAS is similarly lacking in co-occurrence data and is inconsistent with EPA guidance. Furthermore, the proposed regulation of these compounds concurrently with the preliminary determination is beyond EPA's authority under SDWA. EPA should (i) not finalize the preliminary determinations, (ii) evaluate and, if appropriate, re-issue the preliminary determinations following the availability of UCMR 5 data, and (iii) withdraw and re-issue a scientifically sound and adequately supported proposal to regulate the PFAS among these four which EPA can provide a sound basis for a positive determination to regulate.

With respect to the proposed drinking water standards for PFOA and PFOS, the docket does not support finalizing a rule, particularly a rule where attributable benefits outweigh quantifiable costs. The affordability of EPA's rule options is especially questionable for households served by small systems – systems which SDWA requires EPA give particular attention to in crafting a drinking water standard. If EPA moves forward with a final rule, setting MCLs for PFOA and PFOS at 10 ppt is the most appropriate

option among the options EPA has analyzed. This option will prioritize water systems with the highest PFAS levels moving forward immediately.

EPA's proposed approach to require monitoring under the Standard Monitoring Framework and the use of the one-half of the MCL as a trigger level is appropriate. However, the use of reporting results below the PQL is inappropriate as it requires water systems to rely on unreliable data to determine monitoring requirements as part of the regulatory requirements.

In summary, EPA is strongly encouraged to consider the impacts of this rule carefully and to ensure that, if finalized, the regulations are feasible, based on the best available public health information, based on accurate cost assessments, and legally defensible. While the agency has a strong interest in expeditious action, it is important to move actions forward meaningfully and in a way that avoids negative consequences that are avoidable. AWWA's recommendations are intended to assist EPA ensure that high-risk water systems are prioritized while also providing EPA with additional time to get better data and make additional sound, defensible risk management decisions. AWWA's recommendations also reflect EPA placing the onus of PFAS risk reduction on polluters rather than communities through the source water protection actions framed in the agency's Strategic Roadmap for PFAS.

16. References

ATSDR, 2018. Agency for Toxic Substances and Disease Registry. Framework for Assessing Health Impacts of Multiple Chemicals and Other Stressors (Update). February 2018. Available at: https://www.atsdr.cdc.gov/interactionprofiles/ip-ga/ipga.pdf

ATSDR, 2021. Agency for Toxic Substances and Disease Registry. Toxicological Profile for Perfluoroalkyls. May 2021. https://www.atsdr.cdc.gov/toxprofiles/tp200.pdf

ATSDR, 2022. Agency for Toxic Substances and Disease Registry. Public Health Assessment Guidance Manual (PHAGM). Accessed May 30, 2023. Available at: https://www.atsdr.cdc.gov/pha-guidance/index.html.

AWWA, 2020a. American Water Works Association. Drinking Water Treatment for PFAS Selection Guide. 2020. Available at: https://engage.awwa.org/PersonifyEbusiness/Bookstore/Product-Details/productId/87142604

AWWA, 2020b. American Water Works Association. Comments on Preliminary Regulatory Determinations for Contaminants on the Fourth Drinking Water Contaminant Candidate List. May 21, 2020. Available at: https://www.regulations.gov/comment/EPA-HQ-OW-2019-0583-0215

AWWA, 2021a. American Water Works Association. AWWA comments for Science Advisory Board PFAS Review Panel Consideration. December 30, 2021. Available at: https://sab.epa.gov/ords/sab/f?p=100:0:2562407936967:APPLICATION_PROCESS=MEETING_FILE:::MM_ID:5921

AWWA, 2021b. American Water Works Association. Improving the Evaluation of Household-Level Affordability in SDWA Rulemaking: New Approaches. April 2021. Available at: https://www.awwa.org/Portals/0/AWWA/Government/ImprovingtheEvaluationofHouseholdLevelAffordabilityinSDWARulemakingNewApproaches.pdf

AWWA, 2021c. American Water Works Association. AWWA PFAS Case Study: Cape Fear Public Utility Authority. January 19, 2021. Available at: https://www.awwa.org/Portals/0/AWWA/ETS/Resources/Technical%20Reports/CFPUA%20Case%20Study%20Report_FINAL.pdf?ver=2021-01-19-095055-317

AWWA, 2022. American Water Works Association. Comments on Designation of Perfluorooctanoic Acid (PFOA) and Perfluorooctanesulfonic Acid (PFOS) as CERCLA Hazardous Substances. November 7, 2022. Available at: https://www.regulations.gov/comment/EPA-HQ-OLEM-2019-0341-0543

AWWA, 2023a. American Water Works Association. Comments on Proposed 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. February 3, 2023. Available at: https://www.regulations.gov/comment/EPA-HQ-TRI-2022-0270-0068

AWWA, 2023b. American Water Works Association. AWWA Survey Titled: “PFAS Treatment Costs”. Spring 2023. Survey Results Available in Appendix D.

Barger, 2023. Tom Barger, Water Quality Manager for South Central Connecticut Regional Water Authority. E-Mail from Tom Barger Titled "PFAS Cost Estimates – RWA, New Haven, CT". Received May 18, 2023. Available in Appendix E.

Bennington, 2022. Jim Therrien. Pownal water district to get $5.4M treatment system. Bennington Banner. September 26, 2022. Available in Appendix E.

Biden, 2021. President Joseph R. Biden. Protecting Public Health and the Environment and Restoring Science To Tackle the Climate Crisis. January 25, 2021. Available at: https://www.federalregister.gov/documents/2021/01/25/2021-01765/protecting-public-health-and-the-environment-and-restoring-science-to-tackle-the-climate-crisis

Black & Veatch, 2023. Black & Veatch. WITAF 056 Technical Memorandum Update: PFAS National Cost Model Report. May 26, 2023. Available in Appendix B.

Calgon, 2023. Calgon Carbon. Granular Activated Carbon Removes PFOA from Drinking Water: Upstate New York Case Study. Accessed May 30, 2023. Available at: https://www.calgoncarbon.com/app/uploads/Case-Study-GAC-Removes-PFOA-from-Drinking-Water.pdf

California Water Boards, 2023. California Water Boards. PFAS: Per- and Polyfluoroalkyl Substances. Accessed May 25, 2023. Available at: https://www.waterboards.ca.gov/pfas/docs/pfas_monitoring_Q1Q2Q3Q4.xlsx.

CDM Smith, 2018. CDM Smith. Advanced Treatment Options for the Northwest Water Treatment Plant. Prepared for Brunswick County Public Utilities. April 1, 2018. Available at: https://www.brunswickcountync.gov/wp-content/uploads/2018/04/CDM-Smith-Brunswick-Final-Report-April-2018.pdf.

CDPHE, 2023. Colorado Department of Public Health and Environment. PFAS 2020 Sampling Project - Drinking Water Results. Accessed May 25, 2023. Available at: https://docs.google.com/spreadsheets/d/1AiSEOrGgWi1U4owD1hOEKA2q1xTy8NOF/edit#gid=700544691.

Chen et al. 2012. Chen MH, Ha EH, Wen TW, Su YN, Lien GW, Chen CY, Chen PC, Hsieh WS. Perfluorinated compounds in umbilical cord blood and adverse birth outcomes. August 3, 2012. PLoS ONE, 7(8): e42474. doi: 10.1371/journal.pone.0042474. Available in Appendix E.

Circle of Blue, 2019. Circle of Blue. The Price of Water: Water Rates Dashboard. Accessed May 30, 2023. Available at: https://www.circleofblue.org/waterpricing/.

Congress, 2019. 116th U.S. Congress. Public Law 116-92 – National Defense Authorization Act for Fiscal Year 2020. Signed December 20, 2019. Available at: https://www.congress.gov/bill/116th-congress/senate-bill/1790.

Corona, 2022. Corona Environmental Consulting. National Occurrence Database for PFAS in Drinking Water: WITAF #057 Final Summary Report. November 4, 2022. Available in Appendix E.

Darrow et al. 2013. Darrow LA, Stein CR, Steenland K. Serum perfluorooctanoic acid and perfluorooctane sulfonate concentrations in relation to birth outcomes in the Mid-Ohio Valley, 2005-2010.

Environ Health Perspect. October 1, 2013 121(10):1207-13. doi: 10.1289/ehp.1206372. Available in Appendix E.

DOE, 2017. U.S. Department of Energy – Office of Energy Efficiency and Renewable Energy. Water and Wastewater Annual Price Escalation Rates for Selected Cities Across the United States. September 2017. Available at: https://www.energy.gov/femp/articles/water-and-wastewater-annual-price-escalation-rates-selected-cities-across-united.

Eaton et al., 2018. Detailed Analysis of the UCMR 3 Database: Implications for Future Groundwater Monitoring. Journal AWWA. Vol 11, Issue 4, Page 13-25. DOI: 10.5942/jawwa.2018.110.0029. Available in Appendix E.

EFSA, 2018. European Food Safety Authority, Knutsen HK, Alexander J, Barregard L, Bignami M, Bruschweiler B, Ceccatelli S, Cottrill B, Dinovi M, Edler L, Grasl-Kraupp B, Hogstrand C, Hoogenboom LR, Nebbia CS, Oswald IP, Petersen A, Rose M, Roudot AC, Vleminckx C, Vollmer G, Wallace H, Bodin L, Cravedi JP, Halldorsson TI, Haug LS, Johansson N, van Loveren H, Gergelova P, Mackay K, Levorato S, van Manen M, Schwerdtle T. Risk to human health related to the presence of perfluorooctane sulfonic acid and perfluorooctanoic acid in food. December 13, 2018. EFSA Journal, 16(12): e05194. DOI: 10.2903/j.efsa.2018.5194.

EFSA, 2020. European Food Safety Authority, Schrenk D, Bignami M, Bodin L, Chipman JK, Del Mazo J, Grasl-Kraupp B, Hogstrand C, Hoogenboom LR, Leblanc JC, Nebbia CS, Nielsen E, Ntzani E, Petersen A, Sand S, Vleminckx C, Wallace H, Barregard L, Ceccatelli S, Cravedi JP, Halldorsson TI, Haug LS, Johansson N, Knutsen HK, Rose M, Roudot AC, Van Loveren H, Vollmer G, Mackay K, Riolo F, Schwerdtle T. Risk to human health related to the presence of perfluoroalkyl substances in food. September 17, 2020. EFSA Journal, 18(9): e06223. DOI: 10.2903/j.efsa.2020.6223.

ENR, 2023. Engineering News Record. Construction Cost Index History. Access May 30, 2023. Available at: https://www.enr.com/economics/historical_indices/construction_cost_index_history.

EPA, 1986. U.S. Environmental Protection Agency. Guidelines for health risk assessment of chemical mixtures. September 1986. Federal Register 51(185): 34014-34025. Available at: https://www.epa.gov/sites/default/files/2014-11/documents/chem_mix_1986.pdf.

EPA, 2000. U.S. Environmental Protection Agency. Supplementary Guidance for Conducting Health Risk Assessment of Chemical Mixtures. August 2000. Risk Assessment Forum Technical Panel. Washington, DC. EPA/630/R-00/002. Available at https://ordspub.epa.gov/ords/eims/eimscomm.getfile?p_download_id=4486.

EPA, 2005. U.S. Environmental Protection Agency. Economic Analysis for the Final Stage 2 Disinfectants and Disinfection Byproducts Rule. December 2005. Available at: https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P1005OOX.txt.

EPA, 2009. U.S. Environmental Protection Agency. Drinking Water Contaminant Candidate List 3 – Final. October 8, 2009. Federal Register 74, 51850-51862. Available at: https://www.federalregister.gov/documents/2009/10/08/E9-24287/drinking-water-contaminant-candidate-list-3-final.

EPA, 2012. U.S. Environmental Protection Agency. Revisions to the Unregulated Contaminant Monitoring Regulation (UCMR 3) for Public Water Systems. May 2, 2012. Federal Register 77, 26071-26101. Available at: https://www.federalregister.gov/documents/2012/05/02/2012-9978/revisions-to-the-unregulated-contaminant-monitoring-regulation-ucmr-3-for-public-water-systems.

EPA, 2015. U.S. Environmental Protection Agency. Peer Review Handbook, 4th edition. October 2015. Available at: https://www.epa.gov/sites/default/files/2015-10/documents/epa_peer_review_handbook_4th_edition_october_2015.pdf.

EPA, 2019a. U.S. Environmental Protection Agency. EPA's Per- and Polyfluoroalkyl Substances (PFAS) Action Plan. February 2019. Available at: https://www.epa.gov/sites/default/files/2019-02/documents/pfas_action_plan_021319_508compliant_1.pdf.

EPA, 2019b. U.S. Environmental Protection Agency. Preliminary Effluent Guidelines Program Plan 14. October 2019. Available at: https://www.epa.gov/sites/default/files/2019-10/documents/prelim-eg-plan-14_oct-2019.pdf.

EPA, 2020. U.S. Environmental Protection Agency. Interim Guidance on Destroying and Disposing of Certain PFAS and PFAS-Containing Materials That Are Not Consumer Products. December 18, 2020. Available at: https://www.epa.gov/system/files/documents/2021-11/epa-hq-olem-2020-0527-0002_content.pdf.

EPA, 2021a. U.S. Environmental Protection Agency. PFAS Strategic Roadmap: EPA's Commitments to Action 2021 – 2024. October 2021. https://www.epa.gov/system/files/documents/2021-10/pfas-roadmap_final-508.pdf.

EPA, 2021b. U.S. Environmental Protection Agency. Effluent Guidelines Program Plan 15. January 31, 2023. Available at: https://www.epa.gov/system/files/documents/2023-01/11143_ELG%20Plan%2015_508.pdf.

EPA, 2021c. U.S. Environmental Protection Agency. Revisions to the Unregulated Contaminant Monitoring Rule (UCMR 5) for Public Water Systems and Announcement of Public Meetings. December 27, 2021. Federal Register 86 73131-73157. Available at: https://www.govinfo.gov/content/pkg/FR-2021-12-27/pdf/2021-27858.pdf.

EPA, 2021d. U.S. Environmental Protection Agency. National Primary Drinking Water Regulations: Lead and Copper Rule Revisions. Federal Register 86, 4198 – 4312. January 15, 2021. Available at: https://www.regulations.gov/document/EPA-HQ-OW-2017-0300-1550.

EPA, 2021e. U.S. Environmental Protection Agency. Announcement of Preliminary Regulatory Determinations for Contaminants on the Fourth Drinking Water Contaminant Candidate List. March 10, 2023. Federal Register 85, 14098-14142. Available at: https://www.federalregister.gov/documents/2020/03/10/2020-04145/announcement-of-preliminary-regulatory-determinations-for-contaminants-on-the-fourth-drinking-water.

EPA, 2022a. U.S. Environmental Protection Agency. Toxic Substances Control Act Reporting and Recordkeeping Requirements for Perfluoroalkyl and Polyfluoroalkyl Substances. June 28, 2021. Federal Register 86, 33926-33966. Available at: https://www.regulations.gov/document/EPA-HQ-OPPT-2020-0549-0001.

EPA, 2022b. U.S. Environmental Protection Agency. 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. December, 5, 2022. Federal Register 87, 74739-74387. Available at: https://www.govinfo.gov/content/pkg/FR-2022-12-05/pdf/2022-26022.pdf.

EPA, 2022c. U.S. Environmental Protection Agency. Listing of PFOA, PFOS, PFBS, and GenX as Resource Conservation and Recovery Act (RCRA) Hazardous Constituents. Accessed May 30, 2023. Available at: https://www.reginfo.gov/public/do/eAgendaViewRule?pubId=202210&RIN=2050-AH26.

EPA, 2022d. U.S. Environmental Protection Agency. Addressing PFAS Discharges in EPA-Issued NPDES Permits and Expectations Where EPA is the Pretreatment Control Authority. April 28, 2022. Available at: https://www.epa.gov/system/files/documents/2022-04/npdes_pfas-memo.pdf.

EPA, 2022e. U.S. Environmental Protection Agency. Addressing PFAS Discharges in NPDES Permits and Through the Pretreatment Program and Monitoring Programs. December 5, 2012. Available at: https://www.epa.gov/system/files/documents/2022-12/NPDES_PFAS_State%20Memo_December_2022.pdf

EPA, 2022f. U.S. Environmental Protection Agency. Drinking Water Health Advisory: Hexafluoropropylene Oxide (HFPO) Dimer Acid (CASRN 13252-13-6) and HFPO Dimer Acid Ammonium Salt (CASRN 62037-80-3), Also Known as "GenX Chemicals". June 2022. Available at: https://www.epa.gov/system/files/documents/2022-06/drinking-water-genx-2022.pdf.

EPA, 2022g. U.S. Environmental Protection Agency. Drinking Water Health Advisory: Perfluorobutane Sulfonic Acid (CASRN 375-73-5) and Related Compound Potassium Perfluorobutane Sulfonate (CASRN 29420-49-3). June 2022. Available at: https://www.epa.gov/system/files/documents/2022-06/drinking-water-pfbs-2022.pdf.

EPA, 2022h. U.S. Environmental Protection Agency. EPA External Review Draft of Report on the Social Cost of Greenhouse Gases: Estimates Incorporating Recent Scientific Advances. September 2022. Available at: https://www.epa.gov/system/files/documents/2022-11/epa_scghg_report_draft_0.pdf.

EPA, 2022i. U.S. Environmental Protection Agency. Addressing PFOA and PFOS in the Environment: Potential Future Regulation Pursuant to the Comprehensive Environmental Response, Compensation, and Liability Act and the Resource Conservation and Recovery Act. September 7, 2022. Federal Register 87, 54415-54442.

EPA, 2023a. U.S. Environmental Protection Agency. Water Sector Workforce. Accessed May 30, 2023. Available at: https://www.epa.gov/sustainable-water-infrastructure/water-sector-workforce.

EPA, 2023b. U.S. Environmental Protection Agency. IRIS Program Outlook. February 27, 2023. Available at: https://www.epa.gov/system/files/documents/2023-02/IRIS%20Program%20Outlook_Feb%202023.pdf.

EPA, 2023c. U.S. Environmental Protection Agency. EPA takes first-ever federal Clean Water Act enforcement action to address PFAS discharges at Washington Works facility near Parkersburg,

W. Va. April 26, 2023. Available at: https://www.epa.gov/newsreleases/epa-takes-first-ever-federal-clean-water-act-enforcement-action-address-pfas.

EPA, 2023d. TRI Explorer. Accessed May 30, 2023. Available at: https://enviro.epa.gov/triexplorer/tri_release.chemical.

EPA, 2023e. U.S. Environmental Protection Agency. Toxic Substances Control Act Inventory Representation For Products Containing Two Or More Substances: Formulated And Statutory Mixtures. May 2015. Available at: https://www.epa.gov/sites/default/files/2015-05/documents/mixtures.pdf.

EPA, 2023f. U.S. Environmental Protection Agency. 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. April 25, 2023. Federal Register 88, 25080-25205. Available at: https://www.federalregister.gov/documents/2023/04/25/2023-07188/new-source-performance-standards-for-the-synthetic-organic-chemical-manufacturing-industry-and.

EPA, 2023g. U.S. Environmental Protection Agency. Regulatory Impact Analysis for the Proposed New Source Performance Standards for Greenhouse Gas Emissions from New, Modified, and Reconstructed Fossil Fuel-Fired Electric Generating Units. May 23, 2023. Federal Register 88, 33240-33420. Available at: https://www.federalregister.gov/documents/2023/05/23/2023-10141/new-source-performance-standards-for-greenhouse-gas-emissions-from-new-modified-and-reconstructed.

EPA, 2023h. U.S. Environmental Protection Agency. Addressing PFAS in the Environment. April 13, 2023. Federal Register 88, 22399-22403. Available at: https://www.federalregister.gov/documents/2023/04/13/2023-07535/addressing-pfas-in-the-environment.

HDR, 2023. HDR. Well No. 4 PFAS System. Accessed May 30, 2023. Available in Appendix E.

He, 2012. Katherine He. A Calculation of the Environmental Footprint of a Granular Activated Carbon Regeneration Facility. Spring 2012. Available at: https://nature.berkeley.edu/classes/es196/projects/2012final/HeK_2012.pdf

Horsham, 2023a. Horsham Water & Sewer Authority. Scope and Opinion of Probably Cost Replacement of HWSA Well 6. November 11, 2022. Available in Appendix E.

Horsham, 2023b. Horsham Water & Sewer Authority. Comments on Proposed Rule for Per- and Polyfluoroalkyl Substances National Primary Drinking Water Regulation. Submitted May 30, 2023. Available in Appendix E.

Kleinfelder, 2023. Ben Powers, Kleinfelder. PFAS Treatment in New England: A Regional Survey. Presented to NEWWA Spring Conference. March 31, 2023. Available in Appendix E.

LWD, 2023. Lakewood Water District. Per- and Polyfluoroalkyl Substances (PFAS): Perfluorooctanoic acid (PFOA) and Perfluorooctanesulfonic acid (PFOS) National Primary Drinking Water Regulation Rulemaking, Docket No. EPA-HQ-OW-2022-0114. May 24, 2023. Available in Appendix E.

Moody, 2023. Mr. Chris Moody. E-Mail to Mr. Ryan Albert "Re: Call on Tuesday". April 17, 2023. Available in Appendix E.

Mortenson, 2023. Mortenson. Overall Construction Cost Index Q1 2023. Accessed May 30, 2023. Available at: https://www.mortenson.com/cost-index.

NCDEQ, 2022. North Carolina Department of Environmental Quality. Final NPDES Permit. September 15, 2022. Available at: https://www.deq.nc.gov/genx/chemours-npdes-permit-90004209152022/download?attachment?attachment?attachment.

Nolan et al. 2009. Nolan LA, Nolan JM, Shofer FS, Rodway NV, Emmett EA. June 2009. The relationship between birth weight, gestational age and perfluorooctanoic acid (PFOA)-contaminated public drinking water. Reproductive Toxicology, 27: 231-238. DOI: 10.1016/j.reprotox.2008.11.001. Available in Appendix E.

Ohio EPA, 2023. PFAS Sampling Results. Downloaded May 25, 2023. Available at: https://data-oepa.opendata.arcgis.com/datasets/pfas-sampling-results/explore.

PA DEP, 2023. Pennsylvania Department of Environmental Protection. Summary of Results for SDW Sampling Project using EPA Method 537.1. May 2021. Available at: https://files.dep.state.pa.us/Water/DrinkingWater/Perfluorinated%20Chemicals/SamplingResults/PFAS_Sampling_Final_Results_May_2021.pdf.

PFAS Project, 2023. The Project Project lab. Issaquah, Washington. Accessed May 30, 2023. Available in Appendix E.

PNG, 2023. Policy Navigation Group. Benefit-Cost Analysis of EPA's Proposed Per- And Polyfluoroalkyl Substances National Primary Drinking Water Regulation. May 2023. Available in Appendix E.

Ramboll, 2023. Ramboll U.S. Consulting. Comments on the Proposed National Primary Drinking Water Regulation for Per- and Polyfluoroalkyl Substances (PFAS). May 2023. Available in Appendix A.

Rosenfeldt, 2021. Erik Rosenfeldt. Experiences in PFAS Cost of Treatment. July 24, 2021. Available at: https://townhall.virginia.gov/L/GetFile.cfm?File=Meeting%5C58%5C32773%5CMinutes_VDH_32773_v1.pdf.

SAB, 2022. Science Advisory Board to the U.S. Environmental Protection Agency. Transmittal of the Science Advisory Board Report titled, "Review of EPA's Analyses to Support EPA's National Primary Drinking Water Rulemaking for PFAS". August 22, 2022. Report Number: EPA-SAB-22-008. Available at: https://sab.epa.gov/ords/sab/f?p=100:0:3036946749674:APPLICATION_PROCESS=REPORT_DOC::REPORT_ID:1105.

Savitz et al. 2012a. Savitz DA, Stein CR, Bartell SM, Elston B, Gong J, Shin HM, Wellenius GA. Perfluorooctanoic acid exposure and pregnancy outcome in a highly exposed community. May 2012. Epidemiology, 23(3): 386-392. doi: 10.1097/EDE.0b013e31824cb93b. Available in Appendix E.

Savitz et al., 2012b. Savitz DA, Stein CR, Elston B, Wellenius GA, Bartell SM, Shin HM, Vieira VM, Fletcher T. Relationship of perfluorooctanoic acid exposure to pregnancy outcome based on birth records

in the Mid-Ohio Valley. August 1, 2012. Environmental Health Perspectives, 120(8): 1201-1207. DOI: 10.1289/ehp.1104752. Available in Appendix E.

Stein et al. 2009. Stein CR, Savitz DA, Dougan M. Serum levels of perfluorooctanoic acid and perfluorooctane sulfonate and pregnancy outcome. August 19, 2009. American Journal of Epidemiology, 170(7): 837-846. DOI: 10.1093/aje/kwp212. Available in Appendix E.

Strand, 2023. Strand Associates. Water Treatment Plant: Polyfluorlalkyl Substances (PFAS) Removal Study. January 2023. Available in Appendix E.

Tighe & Bond, 2022. MassDEP SRF Program Project Evaluation Form (PEF) PFAS Water Treatment Plants and Meter System Upgrades, Prepared for Webster Water Department. August 2022. Available at: https://www.webster-ma.gov/DocumentCenter/View/14605/PFAS-Project-Narrative-Webster-Water-2316000?bidId=.

Turner Construction, 2023. Turner Construction. Turner Building Cost Index. April 20, 2023. Available at: https://www.turnerconstruction.com/download-document/CostIndex2023Qtr1.pdf.

USBR, 2023. United States Bureau of Reclamation. Bureau of Reclamation Construction Cost Trends. March 3, 2023. Available at: https://www.usbr.gov/tsc/techreferences/mands/cct-pdfs/cct20-23.pdf.

VTDEC, 2023. Vermont Department of Environmental Conservation. PFAS Data. Accessed May 25, 2023. Available in Appendix E.

Walczyk, 2023. E-Mail from Carol Walczyk Titled “PFAS Costs”. Received May 16, 2023. Available in Appendix E.

Wu et al. 2012. Wu K, Xu X, Peng L, Liu J, Guo Y, Huo X. 2012. Association between maternal exposure to perfluorooctanoic acid (PFOA) from electronic waste recycling and neonatal outcomes. Environment International, 48: 1-8. doi: 10.1016/j.envint.2012.06.018. Available in Appendix E.

Appendix A

Detailed Technical Comments on PFAS Toxicological Assessments

Intended for
American Water Works Association (AWWA)

Document type
Report

Date
May 2023

Comments on the Proposed National Primary Drinking Water Regulation for Per- and Polyfluoroalkyl Substances (PFAS)

RAMBOLL

Bright ideas.
Sustainable change.

Contents

1. Introduction

This document provides comments on the science considered and technical methods and approaches applied in the development of the US Environmental Protection Agency's (USEPA) proposed National Primary Drinking Water Regulation (NPDWR) for six per- and polyfluoroalkyl substances (PFAS) including perfluorooctanoic acid (PFOA), perfluorooctane sulfonic acid (PFOS), perfluorononanoic acid (PFNA), hexafluoropropylene oxide dimer acid (HFPO-DA and its ammonium salt, commonly known as GenX Chemicals), perfluorohexane sulfonic acid (PFHxS), and perfluorobutane sulfonic acid (PFBS). Our review for these comments includes the documentation of these methods and approaches in the following public comment drafts:

- Toxicity Assessment and Proposed Maximum Contaminant Level Goal for Perfluorooctanoic Ace (PFOA) in Drinking Water (USEPA 2023a) and Appendices (USEPA 2023b);
- Toxicity Assessment and Proposed Maximum Contaminant Level Goal for Perfluorooctane Sulfonic Acid (PFOS) in Drinking Water (USEPA 2023c) and Appendices (USEPA 2023d);
- Economic Analysis for the Proposed Per- and Polyfluoroalkyl Substances National Primary Drinking Water Regulation (USEPA 2023e) and Appendices (USEPA 2023f);
- Maximum Contaminant Level Goal (MCLG) Summary Document for a Mixture of Four Per- and Polyfluoroalkyl Substances (PFAS): HFPO-DA and its Ammonium Salt (also known as GenX Chemicals), PFBS, PFNA, and PFHxS (USEPA 2023g); and
- Framework for Estimating Noncancer Health Risks Associated with Mixtures of Per- and Polyfluoroalkyl Substances (PFAS) (USEPA 2023h).

These public comment drafts also include the USEPA's attempts to respond the Scientific Advisory Board's detailed comments and review (August 2022) of the following four draft documents:

- Proposed Approaches to the Derivation of a Draft Maximum Contaminant Level Goal for Perfluorooctanoic Acid (PFOA) (CASRN 335-67-1) in Drinking Water (December 2021) (USEPA 2021a);
- Proposed Approaches to the Derivation of a Draft Maximum Contaminant Level Goal for Perfluorooctane Sulfonic Acid (PFOS) (CASRN 1763-23-1) in Drinking Water (December 2021) (USEPA 2021b);
- Draft Framework for Estimating Noncancer Health Risks Associated with Mixtures of Per- and Polyfluoroalkyl Substances (PFAS) (December 2021) (USEPA 2021c); and
- Analysis of Cardiovascular Disease Risk Reduction as a Result of Reduced PFOA and PFOS Exposure in Drinking Water (December 2021) (USEPA 2021d).

The USEPA made substantial revisions and added new material, which is now included in the public comment drafts, to address the extensive comments made by the Science Advisory Board (SAB). In addition, the USEPA responded to the SAB comments in the following report:

- USEPA Response to Final Science Advisory Board Recommendations (August 2022) on Four Draft Support Documents for the USEPA's Proposed PFAS National Primary Drinking Water Regulation (USEPA 2023i).

The revisions to the 2023 public comment drafts include the following, none of which have been peer-reviewed by the SAB, and therefore have to be peer-reviewed during this 60 day public comment period (which will end on May 30, 2023):

- A review of mechanistic data, which was lacking in the December 2021 draft documents, and synthesis of mechanistic data with the animal and human data. The SAB specifically stated *"(US)EPA should include an evaluation of mechanistic/mode of action data for*

those effects considered as the potential basis for the reference doses (RfDs) and cancer slope factors (CSFs)." (p. 2 of introductory letter, USEPA 2022a);

- Newly derived candidate RfDs for total cholesterol are now included in the proposed MCLG documents to align with the cardiovascular disease (CVD) benefits analysis. The SAB specifically stated *"(US)EPA should ensure that recommendations for the draft MCLG documents relating to evidence identification and synthesis are applied to the CVD endpoint."* (p. 4 of introductory letter, USEPA 2022a);
- USEPA quantified benefits of changes in high-density lipoprotein cholesterol (HDLC) in relation to PFOA and PFOS;
- USEPA quantified benefits of changes in elevated blood pressure in relation to PFOS (although this had not requested by the SAB);
- Addition of quantified benefits of birth weight associated with reductions in PFOA/PFOS (SAB requested that USEPA consider risk reduction for additional endpoints);
- Newly derived candidate RfDs for decreases in birth weight (which is sometimes described as low birth weight) in relation to PFOA and PFOS, which were needed to align with quantified birth weight benefits. However, low birth weight has a specific definition (i.e. birth weight below 2500 grams), and this is not the endpoint for which the RfD is derived. The tiered risk assessment approach described in the December 2021 Draft *Framework for Estimating Noncancer Health Risks Associated with Mixtures of Per- and Polyfluoroalkyl Substances (PFAS)* was replaced with a data-driven/menu-based framework for the selection of component-based approaches for PFAS mixture assessment, and the interpretation of any approach as being "screening" or preliminary was minimized. The SAB specifically stated, "*Methods analogous to those classified by USEPA as 'Screening Level' or 'Tier 1' in the framework are potentially being used by states in a decision-making capacity. Issuance of this framework without recognition of that fact may create confusion for public water supplies and risk communication challenges for the public.*" (p. 93-94, USEPA SAB 2022a).

The above changes to the underlying documents and assessments are not exhaustive; however, the comments below are largely focused on a review of these proposed changes.

The proposed approach for the regulation of these compounds includes the development of a hazard index (HI) that is used to determine if the combined levels of 4 PFAS pose a potential health risk. The use of this approach represents the first time a HI approach has been applied for a federal regulation; it has traditionally been applied as a screening tool to make initial decisions regarding chemical remediation. It is also important to note that the approach used by the agency in this case is inconsistent with existing regulatory guidelines; therefore, one of the main focuses of these comments are around the application of a HI outside of a screening approach and the challenges in finding support for this approach in the available science for PFAS.

As the proposed PFAS NPDWR is a significant regulatory action that was submitted to the Office of Management and Budget (OMB) for review, an economic analysis is required under Executive Order 12866. The remaining comments focus on the USEPA's use of the available science related to PFAS exposure and selected endpoints, specifically low birth weight and cardiovascular disease (CVD) to attempt to demonstrate quantifiable and nonquantifiable health risk reduction benefits are likely to occur as the result of compliance with the proposed NPDWR.

2. Comments on the Hazard Index Approach

USEPA has proposed the use of a General HI approach for regulation of drinking water concentrations of four PFAS: PFNA, PFHxS, PFBS, and GenX (USEPA 2023g, p.8-18). However,

rather than following the recommendation of the SAB (USEPA 2022a) to conduct the assessment on the basis of common outcome, the Hazard Index for these 4 compounds are calculated using critical effects (RfDs or MRLs) that are based on different endpoints and target tissues:

- PFNA: Delayed development in mouse offspring.
- PFHxS: Thyroid follicular epithelial hypertrophy/hyperplasia in parental male rats.
- PFBS: Decreased thyroxine in mouse offspring.
- GenX: Liver toxicity in female rat dams.

USEPA attempts to defend this approach using arguments that are inaccurate and contradictory. Initially, USEPA tries to make a broad claim that the diverse effects of PFAS are all associated with a common mechanism involving disruption of cellular signaling:

> *"PFAS, including HFPO-DA, PFBS, PFNA, and PFHxS, 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, and immune and liver function (ATSDR 2021; EFSA 2018, 2020; USEPA 2022c)." (USEPA 2023g, p.2)*

However, the USEPA suggestion that PFAS causes common disruption of biological pathway signaling that results in common adverse effects is essentially an argument for a common mode of action. This argument is difficult to support, given the agency's determination that information to support a common mode of action for PFAS is inadequate:

> *"Because PFAS are an emerging chemical class of note for toxicological evaluations and human health risk assessment, mode of action (MOA) data may be limited or not available for many PFAS." (USEPA 2023g, p.3).*

Moreover, in contradiction to the USEPA suggestion of a common mode of action across all PFAS, the agency concluded in the USEPA Toxicity Assessment for GenX (USEPA 2021a), one of the 4 chemicals included in the HI, that the liver effects of GenX are not consistent with there being a common mode of action for all PFAS:

> *"Although there is evidence for a PPARα MOA in the liver, particularly in the high-dose groups in the available studies, data indicate that liver toxicity extends beyond a single PPARα-based MOA." (USEPA 2021a, p.84).*

USEPA then makes a case for an assumption of dose additivity based on common outcome rather than common mode of action (USEPA 2023g, p.3), citing the USEPA (2000) mixtures guidance. Inexplicably, the USEPA applied a General HI approach across different outcomes for each chemical, apparently based on their assertion of a common mode of action (above), and despite the fact that the multi-outcome approach clearly ignores the recommendation of the SAB (USEPA 2022a). It is also inconsistent with existing USEPA guidelines.

USEPA Mixtures Guidance (1986) does not support the use of dissimilar effects in a Hazard Index:

> *"Since the assumption of dose addition is most properly applied to compounds that induce the same effect by similar modes of action, a separate hazard index should be generated for each end point of concern. Dose addition for dissimilar effects does not have strong scientific support, and, if done, should be justified on a case-by-case basis in terms of biological plausibility." (USEPA 1986, p.9)*

The USEPA Risk Assessment Forum (2000) Mixtures Guidance clearly indicates that the use of critical effects from multiple tissues in a Hazard Index is generally inappropriate. The guidance describes the HI method only in terms of similarity in target organ:

> *"The Hazard Index method has weaker assumptions and data requirements, is more generally applicable, and has more uncertainty in the resulting assessment. Instead*

> *of requiring knowledge of similar mode of action, the Hazard Index method requires only similarity in target organ." (USEPA 2000, p.71)*
>
> *"One of the key desirable features is the constraint to use only data on the effect of concern. Because the Hazard Index is tied to a specific effect, the underlying data should be on that effect. Substituting data on the critical effect introduces an unknown degree of conservatism, so that the Hazard Index is inflated by an unknown amount." (USEPA 2000, p.85)*
>
> *"The use of an acceptable level in the relative toxicity scaling factor (e.g. 1/RfD) may be overly health protective in that the RfD (or RfC) is based on the critical effect, defined as the toxic effect occurring at the lowest dose. When the Hazard Index is calculated for some different, less sensitive effect, the RfD will be too low, so the factor (1/RfD) will overestimate the relative toxicity and the Hazard Index will be too large. One alternative that avoids this critical effect conservatism is to use a toxicity-based exposure level that is specific to the target organ of interest and is derived similarly to an RfD (or RfC). For oral exposures, this value is called the target organ toxicity dose or TTD (Mumtaz et al., 1997)." (USEPA 2000, p.82)*

Indeed, the use of a Hazard Index based on the combination of endpoints from multiple target tissues, apart from screening purposes, is not supported by any national risk assessment agency. It is inconsistent with ATSDR guidance.

The ATSDR (2022) Public Health Assessment Guidance Manual indicates that when "the health guideline for each contaminant is based on different target organs, health assessors will need to calculate a target-organ-specific HQ for each contaminant. These target-organ HQs can now be added together to give a Tier 3 HI based on the same target organ." (ATSDR 2022, p.8)

The ATSDR Framework for Assessing Health Impacts of Multiple Chemicals and Other Stressors (2018) indicates that the use of different target organ toxicities is reserved for screening:

> *"Because it is based on the assumption of dose additivity, the hazard index method is most appropriately applied to components that cause the same effect by the same mechanism or mode of action. In practice, it may be applied to components with different target organs as a screening measure." (ATSDR 2018, p.43)*

USEPA (2023g) does not provide any justification for failing to apply the Target-organ Toxicity Dose (TTD) methodology that ATSDR has developed specifically to address situations where there is an overlap in effects across a mixture of chemicals, but where the critical effects are different: "A Target-organ Toxicity Dose (TTD) for each end point of concern is calculated using appropriate MRL (or RfD) methodology, and then used in estimating the end-point specific HQs and hazard indices." (ATSDR 2018, p.45)

Use of the ATSDR methodology is particularly important in the case of this NPDWR, since the General HI used by the agency is based on the critical (lowest) effect for each chemical, regardless of target tissue. As pointed out by ATSDR, a HI based on TTDs will certainly be higher than one based on the General HI, indicating that the currently proposed General HI approach is overly conservative.

The draft mixtures framework (USEPA 2021b) referred to the HI based on multiple target tissues as a "Screening-Level HI" to differentiate it from a Target Organ Specific Hazard Index (TOSHI), which they indicated would be more consistent with the USEPA (2000) mixtures guidelines. The SAB specifically supported the USEPA Framework's use of the TOSHI rather than the Screening Level HI:

> *"The SAB supports dose additivity based on a common outcome, instead of a common mode of action as a health protective default assumption and does not propose another default approach."* (USEPA 2022a, p.90)

However, the SAB indicated that the agency should avoid referring to the multiple target tissue HI approach as a "Screening-Level HI", to avoid the appearance of disparaging the work of states that have been using it in their regulations:

> *"Methods analogous to those classified by USEPA as 'Screening Level' or 'Tier 1' in the framework are potentially being used by states in a decision-making capacity. Issuance of this framework without recognition of that fact may create confusion for public water supplies and risk communication challenges for the public."* (USEPA 2022a, p.3-94)

In response to this SAB concern, the revised Framework (USEPA 2023h) refers to the Screening-Level HI as a "General HI". USEPA then applied the General HI approach rather than the TOSHI approach in deriving the MCLG for mixtures of 4 PFAS (USEPA 2023g). However, as pointed out above, the USEPA SAB (2022a) had clearly indicated that the USEPA should base their assessment on common outcome, which would require the use of the TOSHI approach.

2.1 Conclusions and Recommendations

- The proposed use of a HI based on different target organs or endpoints for estimation of a regulatory value has no support in existing agency guidelines or those of other national and international authoritative bodies.
- The agency should delay promulgating a HI-based assessment until they have developed the necessary Target Tissue Doses (TTDs) to support the use of the TOSHI approach.
- The TTDs can readily be derived using the existing ATSDR methodology (ATSDR 2018).

3. Comments on the Birth Weight Risk Reduction Analysis

In the 2021 documents, the USEPA had not quantified benefits of birth weight risk reductions associated with reductions in exposure to PFOA and PFOS in drinking water. The SAB had recommended that the USEPA consider risk reduction analyses for other endpoints, provided a sufficient rationale existed. The quantified health benefits associated with birth weight impacts now include the following, which have not been peer-reviewed by the SAB:

- Increase in birth weight (in millions of grams); and
- Number of birth-weight related deaths avoided.

Under the Proposed Option (MCL of 4 ppt for PFOA and 4.0 ppt for PFOS and an HI of 1.0 for PFNA, HFPO-DA (GenX chemicals), PFHxS, and PFBS), the USEPA calculated an expected increase in birth weight of 209,300,000 grams and 1,232.7 birth-weight related deaths avoided when the Proposed Option was compared to baseline drinking water concentrations.

Previously, after integrating the evidence, the USEPA had been unclear regarding its strength of evidence conclusion that PFOA and PFOS are associated with low birth weight or decreases in birth weight. Nevertheless, the USEPA (2021a, 2021b) had derived candidate RfDs for decreases in birth weight based on epidemiological studies before selecting a critical effect with the lowest point of departure human equivalent dose (which was for vaccine response). The SAB (USEPA 2022a) requested that the "US*EPA consider reevaluating its strength of evidence conclusions for some human endpoints, including (but not necessarily limited to) decreased immune response, increased liver enzymes, increased serum lipids (for PFOA) and decreased fetal growth to determine if they are better described as having "likely" or "strong" evidence rather than "suggestive" or "moderate" evidence of an association with exposure to PFOA/PFOS.* ***Such a***

reevaluation should consider studies included in the 2016 HESD and more recent studies published after the end date of the literature search for the current draft." (emphasis added, p. 23, USEPA 2022a).

Most recently, the USEPA (2023a, 2023c) judged the evidence of an association between PFOA or PFOS and fetal growth restriction as "likely" based on "moderate" evidence in humans (Note: The evidence integration included a review of mechanistic data which had not been reviewed previously by the SAB. There were also new figures that had not been reviewed previously, including the forest plots for low birth weight and small for gestational age, which had been omitted from the USEPA 2021a, 2021b documents). This conclusion of a likely association between PFOA or PFOS and fetal growth restriction based on moderate evidence in humans also allowed for the justification of quantified health benefits from birth weight impacts associated with the Proposed Option (as well as other regulatory options).

3.1 Candidate Reference Dose for Low Birth Weight

For PFOA, the USEPA derived a Reference Dose (RfD) for low birth weight (LBW, defined as birth weight<2500 grams) using a hybrid approach for defining the benchmark response (BMR), where the adverse health effect (LBW) was estimated using the dose that increases the percent of responses falling below the clinical definition of LBW (<2500 g). In 2018, 8.27% of live births fell below 2500 g (CDC 2023a as reported in USEPA 2023b and 2023d). As a result, the USEPA selected a BMR of 5% and the background response of 8.27% to calculate a dose that results in 12.86% of the responses falling within a clinical definition of low birth weight (<2500 g). For the dose-response association, the USEPA (2023a) chose the coefficient for the effect of PFOA on decreased birth weight from Wikstrom et al. (2020) (β -68.0 g per ln-ng/mL, 95% CI -112.0 to -24.0). When re-expressed to ng/mL (which was used to estimate the benchmark dose (BMD) and the lower bound on the BMD (BMDL)), β was -41.0 per ng/mL, 95% CI -67.5 to -14.5 g per ng/mL.

The use of LBW as the critical effect is inconsistent with other regulatory agencies that found small decreases in birth weight in relation to PFOA and PFOS but not increased risk of low birth weight. These include the following examples:

- ATSDR (2021) reported *"Small (<20-g or 0.7-ounce decrease in birth weight per 1 ng/mL increase in either PFOA or PFOS blood level) decreases in birth weight (PFOA, PFOS)."*
- ATSDR (2021) also reported "*most studies found no association between maternal serum PFOA levels and the risk of low birth weight infants (typically defined as <2,500 g) (Chen et al. 2012a; Darrow et al. 2013; Fei et al. 2007, 2008a; Manzano-Salgado et al. 2017a; Savitz et al. 2012b; Stein et al. 2009) or found a decreased risk of low birth weight infants (Nolan et al. 2009; Savitz et al. 2012a). Similarly, most studies found no increases in the risk for small for gestational age (Chen et al. 2012a; Fei et al. 2007, 2008a; Hamm et al. 2010; Lauritzen et al. 2017; Manzano-Salgado et al. 2017a; Savitz et al. 2012b; Wang et al. 2016; Whitworth et al. 2012a)*." (p. 465)
- EFSA (2020) reported that PFOA or PFOS exposure was associated with *"reduced birth weight."* Furthermore, EFSA (2018, 2020) concluded that the decrease in birth weight is small after adjusting for confounders and "*the potential longer term consequences of this decrease are unclear."*
- EFSA (2020) also stated "*As already explained in the previous Opinion on PFOA and PFOS (EFSA CONTAM Panel, 2018), the association with reduced birth weight might at least partly be explained by changes in the physiology during pregnancy, although a recent study seemed to strengthen the causality of the effect (Meng et al., 2018; see also Section 3.3.4.1.1). The remaining decrease in birth weight after adjusting for confounders was not large and the potential longer term consequences of this decrease*

> *are unclear. Thus far, there is little evidence for an increase in the proportion of children with low birth weight (< 2,500 g).”* (p. 138)

In the hazard assessment, the USEPA (2023a, 2023c) evaluated and integrated evidence for fetal growth restriction by combining epidemiological studies on the risk of LBW (birth weight<2500 grams) with studies on the risk of small for gestational age (SGA, primarily defined in epidemiological studies as birth weight below the 10th percentile for the gestational age). Although these endpoints can be correlated, they are *not* equivalent endpoints and they should not be evaluated as if they are same. The forest plots (figures 3-54, 3-55, 3-56, and 3-57, USEPA 2023a) are new and were not included in the 2021 draft documents. The USEPA (2023a) concluded that the evidence supports increased risk of LBW and SGA in relation to PFOA:

> *“Overall, nine of the eleven informative studies reporting main effects for either SGA or LBW or both showed some increased risks with increasing PFOA exposures. The magnitude of the associations was typically from 1.2 to 2.8 with limited evidence of exposure-response relationships among the studies with categorical data. Although the number of studies was fairly small, few discernible patterns across study characteristics or confidence ratings were evident across the SGA or LBW findings. For example, four of the nine studies showing increased odds of either SGA or LBW were based on early sampling biomarkers. Collectively, the majority of SGA and LBW studies were supportive of an increased risk with increasing PFOA exposures.” (p. 3-212, USEPA 2023a).*

The tables and figures in the PFOA toxicity assessment (USEPA 2023a) reported that eight studies were informative for a total of nine results (Chu et al. 2020, Wang et al. 2016, Wikström et al. 2020, Lauritzen et al. 2018, Manzano-Salgado et al. 2017, Govarts et al. 2016, Hjermitslev et al. 2020, Meng et al. 2018). Closer inspection showed that the results were not consistent within or between studies. For example:

- Three studies stratified results according to sex. One study found increased risk of LBW in girls, but not boys (Wikström et al. 2020), one study found increased risk of SGA in girls and decreased risk of SGA in boys (Wang et al. 2016), and one study found increased risk of LBW in boys, and decreased risks in girls (Manzano-Salgado et al. 2017). Separately, Manzano-Salgado et al. (2017) also reported increased risk of SGA in boys and decreased risks in girls.
- Lauritzen et al. (2018) stratified by country of birth and reported decreased risk of SGA in Norway (median PFOA concentration, 1.62 ng/mL and median PFOS concentration, 9.74 ng/mL) and increased risk of SGA in Sweden (median PFOA concentration, 2.33 ng/mL and median PFOS concentration 16.4 ng/mL). (The range of PFOA and PFOS was similar among study participants from both countries, suggesting other explanations are likely for the differences in risks).
- Exposure-response results were not consistent within studies where exposures were measured in the general population (at low concentrations of PFOA and PFOS):
    - Meng et al. (2018) reported no association between risk of LBW per doubling of PFOA exposure (OR 1.0, 95% CI 0.7–1.5) and slightly increased odds ratios when PFOA exposure was categorized into quartiles and Q2 (OR 1.5, 95% CI 0.8–3.1), Q3 (OR 1.2, 95% CI 0.5–2.5), and Q4 (OR 1.5, 95% CI 0.7–3.3) were compared to Q1.
    - Chu et al. (2020) reported a slightly increased OR for LBW of 1.16 per ng/ml increase of PFOA (median concentration 1.54 ng/mL, interquartile range 0.957 to 2.635 ng/mL), but no increased risks when exposure was categorized into quartiles (OR 1.0 for 4th quartile (≥2.64 ng/ml) compared to 1st quartile (≤0.096 ng/ml)).

- Three studies did not show exposure-response relationships or trends when exposure was categorized into quartiles of exposure (Meng et al 2018; Wikström et al. 2020; Chu et al. 2020). Wikström et al. (2020) showed an increased risk only for PFOA>2.30 ng/mL (4th quartile) compared to <1.1 ng/ml (1st quartile). Chu et al. (2020) did not find any increased risk of LBW when exposure was categorized by quartiles of exposure.
- When stratified by maternal sampling in early pregnancy (1st trimester) and maternal sampling in later pregnancy (2nd 3rd trimesters, cord blood, after delivery), there were 4 studies of sampling in early pregnancy (Hjermitslev et al. 2020; Wikström et al. 2020; Meng et al. 2018; Manzano-Salgado et al. 2017) and 4 studies of sampling in late pregnancy (Chu et al. 2020; Govarts et al. 2016; Wang et al. 2016; Lauritzen et al. 2018). Although Wikström et al. (2020) reported an increased risk of LBW in girls when the highest exposure was compared to the lowest exposure, other studies that sampled early in pregnancy reported decreased risks of LBW (Hjermitslev et al. 2020; Manzano-Salgado et al. 2017).

Importantly, and despite the request by the SAB that the USEPA re-evaluate studies published before and included in the 2016 Health Effects Support Documents (HESD) for PFOA (USEPA 2016a) and PFOS (USEPA 2016b), the USEPA (2023a, 2023c) did not include studies that evaluated risk of LBW or risk of SGA and exposure to PFOA or PFOS that were published prior to 2017 in the overall integration of evidence. Consequently, the USEPA did not consider at least 7 studies that evaluated LBW and did not find an increased risk of low birth weight in relation to PFOA or PFOS (Darrow et al. 2013; Nolan et al. 2009; Savitz et al. 2012a, 2012b; Stein et al. 2009; Chen et al. 2012; Wu et al. 2012). [Note: These studies were summarized in Table D.1.2 in the Appendix to the PFOA Toxicity Assessment (USEPA 2023b) and the study quality evaluation was presented in Figure 3-45 (USEPA 2023a), but these studies were not included in the forest plots (Figures 3-54 and 3-55, USEPA 2023a) or discussed in the integration of evidence]. In contrast, and in response to the request by the SAB (USEPA 2022a), the USEPA (2023a,2023c) had integrated evidence regarding immunotoxicity studies and cholesterol studies that were older and included in the 2016 HESD documents (USEPA 2016a,2016b).

The USEPA concluded in the PFOS toxicity assessment (USEPA 2023c):

> *"Collectively, the majority (7 of 10) of SGA and LBW studies were supportive of an increased risk with increasing PFOS exposures. The increased odds ranged from 1.19 to 4.14 although evidence of exposure-response relationships was lacking. There was no evidence of differences by study confidence as five of these seven were either high (n=4) or medium (n=1) confidence. There was also no evidence of sample timing differences as the majority of studies with associations were reported in studies based on early sampling periods." (p. 3-209, USEPA 2023c).*

The USEPA (2023c) did not provide references within the above sentence; however, review of tables and figures reported the following studies were high confidence (Chu et al. 2020, Manzano-Salgado et al. 2017, Lauritzen et al. 2018, Wikström et al. 2020) or medium confidence (Govarts et al. 2016; Hjermitslev et al. 2020; Meng et al. 2018) despite the following issues:

- Exposure-response relationships were generally not seen.
- Four studies (Manzano Salgado et al. 2017; Wikström et al. 2020; Meng et al. 2018; Hjermitslev et al. 2020) were based on sampling during early pregnancy while three studies (Lauritzen et al. 2018, Govarts et al. 2016, Chu et al. 2020) were based on sampling late in pregnancy.

The USEPA (2023e) conflates decreases in birth weight with low birth weight in the economic analysis. The USEPA provided a rationale for estimating medical costs associated with changes in infant birth weight and the value of avoiding infant mortality at various birth weights by citing to health effects in relation to low birth weight specifically:

> *"LBW 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 et al., 1991; J. R. Behrman et al., 2004; R. E. Behrman et al., 2007; Joyce et al., 2012; Kowlessar et al., 2013; Colaizy et al., 2016; Nicoletti et al., 2018; Klein et al., 2018). Recent literature also linked LBW 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)."* (USEPA 2023e, p. 6-360)

> *"Low birth weight (LBW) is an important health outcome affected by PFOA/PFOS exposure because it is a significant factor in survival rates and medical care costs among infants (ATSDR, 2021)."* (USEPA 2023e, p. 6-13)

> *"Epidemiology studies on PFOA supported an increased risk of LBW in infants with PFOA exposures (USEPA, 2023a). Similarly, epidemiology studies on PFOS showed an increased risk of LBW infants with PFOS exposures. Overall, most epidemiology studies evaluating the association between maternal serum PFOA/PFOS and birth weight reported negative relationships (i.e. increased exposure is associated with decreased birth weight) (Darrow et al., 2013; Verner et al., 2015; Govarts et al., 2016; Negri et al., 2017; Starling et al., 2017; Sagiv et al., 2018; Chu et al., 2020; Dzierlenga et al., 2020; Wikström et al., 2020; Yao et al., 2021). FN30: Recent evidence indicates that relationships between maternal serum PFOA/PFOS and birth weight may be impacted by changes in pregnancy hemodynamics (Sagiv et al., 2018; Steenland et al., 2018)."* (USEPA 2023e, 6.13)

When considering the evidence for risk of low birth weight in relation to PFOA and PFOS, the USEPA combines studies of the risk of LBW with studies of the risk of SGA (USEPA 2023a, 2023c). There is little evidence that the risk of LBW or the risk of SGA is increased (see remarks above).

3.2 Exposure-response functions for PFOA and PFOS and decreases in birth weight used in the Economic Analysis

The USEPA (2023e) Economic Analysis relies on the exposure-response coefficients (slope factors) for decreases in birth weight from the main analyses of a meta-analysis of birth weight effects in relation to PFOA (Steenland et al. 2018) which reported a mean birth weight decrease of 10.5 g per ng/ml (95% CI −16.7, −4.4) and a separate meta-analysis of birth weight effects in relation to PFOS (Dzierlenga et al., 2020) which reported a mean birth weight decrease of 3.0 g per ng/ml (95% CI −4.9, −1.1). [NOTE: An average decrease of 10 grams is equivalent to a decrease of approximately 0.35 ounces]. The biological or clinical significance of such small changes in birth weight is uncertain. The exposure-response function (β -10.5 g birth weight per ng/mL serum) for PFOA used in the economic analysis is also considerably smaller than the coefficient (β -41.0 g per ng/mL, 95% CI −67.5, −14.5 for PFOA) from the study selected for the critical effect and the calculation of the BMD and BMDL (Wikström et al. 2020). For PFOS, the exposure-response function for the economic analysis (β -3.0 g per ng/mL) is slightly smaller than the coefficient (β -8.4 g per ng/mL, 95% CI −16.0, −0.5) used for deriving the candidate RfD for low birth weight based on Wikström et al. 2020).

Both of these meta-analyses (Steenland et al. 2018; Dzierlenga et al. 2020) conducted specific sensitivity analyses to evaluate bias associated with maternal sampling during late pregnancy

compared to maternal sampling during early pregnancy. Both meta-analyses reported that essentially no effect on birth weight was seen when maternal blood is sampled early in pregnancy, while a relatively larger effect on birth weight was seen when maternal blood is sampled late in pregnancy. In general, this suggests that any effect of PFOA or PFOS on birth weight is confounded by the time of sampling (Steenland et al. 2018, 2020; Dzierlenga et al. 2020). In brief, an increased glomerular filtration rate and maternal plasma volume expansion during pregnancy leads to an increased elimination of PFOA and PFOS. Plasma volume expansion and glomerular filtration rate are also related to birth weight. When PFAS in serum is sampled late in pregnancy, the magnitude of the glomerular filtration rate and the plasma volume expansion can distort the association between PFAS and birth weight. Therefore, using the main effect from the meta-analysis (which is essentially an average of birth weight effects reported from early in pregnancy and late in pregnancy) will overestimate the health benefits associated with birth weight risk reductions under the assumption that pregnancy hemodynamics confound the association.

Steenland et al. (2018) found that there was no effect on birth weight after including the C8 Science study in the meta-analysis:

> *"Our meta-analysis including nine new studies, with an almost equal number of births as prior studies, shows a modest inverse association between maternal or cord PFOA and birthweight, with large heterogeneity across studies.* ***The two studies with exposure above background levels showed no association, and similarly, restriction to studies with blood sampling conducted early in pregnancy or shortly before conception showed little or no association. These findings are consistent with confounding and/or reverse causality being responsible for the inverse association seen in studies with low background exposure levels and blood sampling conducted later in pregnancy, when confounding and/or reverse causality are likely to be more important."***

Overall, there is little evidence that PFOA or PFOS at serum concentrations reported in the general population affect developmental outcomes. The USEPA (2023e) confirmed that there was generally a lack of evidence for exposure-response associations between PFOA and PFOS and other development outcomes:

> *"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. As described in Section 6.2, these developmental outcomes have limited epidemiology and toxicology evidence showing associations with PFOA/PFOS exposure and due to this uncertainty, these outcomes were not further assessed." (p. 6-36).*

3.3 Other factors that have affected mean birth weight

The USEPA (2023e) economic analysis calculates that the expected value of birth weight increases, assuming the MCLs are set to 4.0 ppt for PFOA and PFOS plus a hazard index of 1.0 for PFNA, PFBS, PFHxS, and HFPO-DA, is an average increase of 50 grams (1.8 ounces) in mean birth weight. According to US Natality data (CDC 2023a), the mean birth weight in 2018 was 3261.64g and 8.3% of births were low birth weight (<2500 g). If average birth weight were to increase by 50 grams, the mean birth weight would be 3,316.84 g.

During 2003-2018, median PFOS in blood serum decreased substantially by 12 ng/mL, from 14.6 ng/ml in 2003-2004 to 2.6 ng/ml in 2017-2018 (USEPA 2022b) in the population of women aged 16-49 years old (women of childbearing age). During the same years, median PFOA in blood serum also decreased but by a smaller absolute change of 2.1 ng/mL, from 3 ng/mL in 2003-2004 to 0.9 ng/mL in 2017-2018) (USEPA 2022b).

Although the average birth weight in 2003 was 3291.03 grams, which was 30 grams higher than average birth weight in 2018, there were fewer births of low birth weight babies (CDC 2023a). In 2003, 7.9% of births were of low birth weight while in 2018, 8.3% of births were of low birth weight. Together, these data suggest further reductions of PFOA and PFOS concentrations in blood serum are unlikely to result in measurable increases in average birth weight (CDC 2023a).

Tilstra and Masters (2020) reported that average birth weight decreased in the United States since at least 1990. In 1990, average birth weight was 3314.5 grams (approximately 50 grams more than in 2018). However, Tilstra and Masters (2020) provided an analysis that argued that the shift to lower average birth weight is due to changes in obstetric practices (more c-sections and scheduled births). As a result, there are fewer and fewer vaginal births at 40-42 weeks, when babies are heavier; most of these births have shifted to 37-39 weeks because of changes in obstetric practices. This shift has affected the average birth weight.

Data from other areas where PFOS and PFOA are found in the blood serum at similar concentrations to the US also provided evidence that PFOS and PFOA in blood serum at general population levels do not result in decreased birth weight. For example, birth weight in the Faroe Islands (where decreases in antibody response to diphtheria vaccination in relation to increases in PFOA and PFOS form the basis for the RfD for immunotoxicity effects, specifically decreases in vaccine response at age 7 in relation to PFOA or PFOS at age 5) has increased over the past 50 years. For the years 1969-1981, Olsen and Joensen (1985) reported that the average birth weight of liveborn infants delivered in the Faroe Islands was the highest average weight (3,610 g) reported by 33 countries. More recently, Olsen et al. (2023) found that that the mean birthweight in the Faroe Islands was higher than other Nordic countries and had increased during 2010–2019 in the Faroe Islands.

3.4 Conclusions and Recommendations

- When integrating the evidence of birth weight effects and arriving at an evidence stream judgment for humans, the USEPA should consider older studies of LBW in relation to PFOA and PFOS that predated the 2016 HESD (USEPA 2016a, 2016b). Reconsidering these studies likely decreases confidence in the judgment of evidence.
- The USEPA should provide further rationale for using Wikström et al. (2020) as the critical study for the association of decreased birth weight given that it is only one study that sampled PFAS in serum during early pregnancy that showed an association between decreased birth weight and increases in PFAS; other studies that sampled during early pregnancy did not show an association or they showed an attenuated association, which potentially leads to a conclusion that the evidence for an association between PFAS and decreased birth weight is inconsistent after considering potential confounding.
- The USEPA should provide a quantified sensitivity analysis and further discussion of the effects of confounding or reverse causation by pregnancy hemodynamics on the health benefits analysis.
- Available data suggest that further reductions of PFOA and PFOS concentrations in blood serum are unlikely to result in measurable increases in average birth weight. Data from other areas where PFOS and PFOA concentrations are elevated suggest that increases in average birth weight and decreases in infant mortality are likely not expected. For example, PFOS and PFOA in blood serum have been measured in maternal serum in birth cohort studies in the Faroe Islands with mean concentrations similar to that reported in the general population in the US (Grandjean et al. 2012). Studies have reported that the mean birthweight in the Faroe Islands was higher than other Nordic countries and had continued to increase during 2010–2019 in the Faroe Islands (Olsen and Joensen 1985; Olsen et al. 2023). Although there are no published studies of birth weight in relation to PFAS serum concentrations in the Faroe Islands, it is unlikely that small decreases in birth

weight in relation to PFAS – should the association exist in this population – have adverse health consequences.

4. Comments on the Cardiovascular Disease Risk Reduction Analysis

Previously, the USEPA had proposed using the atherosclerotic cardiovascular disease (ASCVD) model to estimate reductions in CVD risks associated with reductions in exposure to PFOA and PFOS in drinking water. The SAB was generally supportive of the overall approach for estimating reductions in CVD risk; however, the SAB noted that the approach did not mesh with the USEPA's conclusion that there was insufficient evidence of increased CVD risk to inform a candidate RfD. In response to SAB feedback, the USEPA (2023a, 2023c) developed RfDs for total cholesterol (TC) as a precursor to CVD and to further justify its use of the ASCVD model – which uses TC as one of several variables to estimate 10-year risk of CVD events – for quantifying CVD benefits. The quantified health benefits now include the following:

- Number of non-fatal myocardial infarction (MI) cases avoided;
- Number of non-fatal ischemic stroke (IS) cases avoided; and
- Number of CVD deaths avoided.

Under the Proposed Option (MCL of 4 ppt for PFOA and 4 ppt for PFOS and an HI of 1.0 for PFNA, HFPO-DA (GenX chemicals), PFHxS, and PFBS), the USEPA calculated that the following morbidity and mortality are avoided when the Proposed Option is compared to baseline drinking water concentrations: 6,081.0 non-fatal MI cases avoided; 8,870.8 non-fatal IS cases avoided; and 3,584.6 CVD deaths avoided (USEPA 2023e).

In order to calculate the avoided CVD related mortality and mortality, the USEPA used exposure-response functions of serum PFOA and PFOS on TC, and serum PFOS (but not serum PFOA) on systolic blood pressure (BP) to estimate annual changes in TC and BP biomarkers.

4.1 Exposure-response functions for PFOA and PFOS and TC

The USEPA (2023f) conducted a meta-analysis of epidemiological studies of the general population of the associations between certain PFAS and total cholesterol to estimate the exposure-response function. Using these exposure-response functions presumes that further reductions in average PFOA and PFOS concentrations in serum in the general population will result in decreases in serum cholesterol, and that decreases in serum cholesterol will lead to decreases in CVD. In other words, this presumes that serum cholesterol is an intermediate variable on the causal pathway between PFAS exposure and CVDs.

The PFOA-TC exposure-response function developed by USEPA (2023f) is the summary estimate from a meta-analysis of four studies of the general population (Nelson et al. 2010; He et al. 2018; Dong et al. 2019; Fan et al. 2020). [NOTE: The slope factor is 1.57 mg/dL per ng/mL serum PFOA. The number of studies is either four (see p. F-11, Table F-2, and p. F-12, Figure 2, USEPA 2023f) or six (see p. F-10, USEPA 2023f and elsewhere in the documents).] Similarly, the PFOS-TC exposure-response function developed by USEPA (2023f) is the summary estimate from a meta-analysis of five studies of the general population (Chateau-Degat et al. 2010, Nelson et al. 2010, He et al. 2018, Dong et al. 2019, Fan et al. 2020), including the same four studies used for the PFOA-TC exposure-response function. Four of these studies were based on cross-sectional analyses serum PFAS and TC from National Health and Nutrition Examination Surveys (NHANES) and included overlapping years. For example:

- Nelson et al. (2010) included NHANES participants from 2003 to 2004; individuals taking cholesterol lowering medication were excluded.

- He et al. (2018) included NHANES participants from 2003-2004 to 2011-2012; individuals taking cholesterol lowering medication were not excluded.
- Dong et al. (2019) included NHANES participants from 2003-2004 to 2013-2014; individuals taking cholesterol lowering medications were excluded.
- Fan et al. (2020) included NHANES participants from 2011-2012 to 2013-2014; individuals taking cholesterol lowering medication were not excluded.

Overall, the USEPA (2023f) identified and included 14 studies in the meta-analysis of exposure-response relationships between PFOA or PFOS and TC. When all 11 studies of PFOA and TC are included in the meta-analysis, the slope factor is 0.003 mg/dL TC per ng/mL in serum PFOA. However, the exposure-response relationship used for the benefits analysis was based only on the summary estimate of the four (PFOA) or five (PFOS) studies that reported linear slope-relationships (beta coefficient for a change in TC or HLD-C in mg/dL to increases in serum PFOA or PFOS in ng/mL). These are the studies of the general population and the coefficient (1.57 mg/dL TC per ng/mL serum PFOA) used in the benefits analysis is 500-fold higher than when all 11 studies are included. This means that the estimated benefits will be much greater using the coefficient from the meta-analysis of the four (overlapping) general population studies than from using the coefficient from the 11 studies of PFOA-TC.

In fact, there appears to be a non-linear association between PFOA and TC, which is not accounted for when a linear slope factor is used over a relatively narrow range of PFOA in serum seen in the general population. In an evaluation of serum lipids in participants in the C8 Health Science study, Steenland et al. (2009) reported that the exposure-response function was steeper at concentrations of total cholesterol (TC) below approximately 208 mg/dL. Predicted TC leveled off at around 50 ng/ml of PFOA. By relying on the exposure-response function from four studies (PFOA) or five studies (PFOS) of the general population (with average PFOA and PFOS serum concentrations below 25 ng/ml), the calculated health benefits are greater than would be expected than if the exposure-response function was based on the distribution of serum PFOA and PFOS seen in occupational populations and studies of communities with drinking water contaminated by PFOA or PFOS. These populations have blood serum concentrations that are 2 to 4 orders of magnitude greater than those in the general population.

4.2 Issues related to use of the ASCVD model

The USEPA assumes that CVD (myocardial infarctions and strokes) can be reduced indirectly by decreasing average serum PFOA and PFOS concentrations further, which would lead to decreased total cholesterol; however, the USEPA has not shown evidence that PFAS exposure (particularly PFOA or PFOS) directly increases the risk of CVD. The USEPA (2023a, 2023c) did not acknowledge that epidemiological studies of PFAS exposures have not observed increased risks of CVD even in studies of populations exposed to the highest concentrations; instead, the USEPA suggested that the results are inconsistent.

The benefits analysis does not directly use PFOA or PFOS concentrations as inputs to the pooled cohort ASCVD model that evaluates the 10-year probability of CVD outcomes (Goff et al. 2014). Instead, the benefits analysis focuses on the exposure-response function between PFOA and PFOS and the precursor endpoint (e.g. total cholesterol) from a meta-analysis of results from epidemiological studies to calculate inputs to the ASCVD model.

Even under the assumption that PFOA or PFOS in serum leads to high total cholesterol, and a shift in the distribution of average cholesterol leads to an increased proportion of individuals with high cholesterol (which potentially affects a large population), there is a lack of evidence that such a shift has occurred based on PFOA and PFOS concentrations in the general population over the past 20 years. As such, there is substantial uncertainty in the population health benefit of

reduced TC by further reducing PFOA and PFOS concentrations from the baseline assumptions to 4 ppt each. The USEPA partially acknowledged this uncertainty when it stated:

> *"The analysis assumes that the CVD risk impact of changes in TC/BP from reductions in serum PFOA/PFOS is the same as the CVD risk impact of changes in these biomarkers due to other reasons such as behavioral changes or medication."* (P. 6-117,6-118, USEPA 2023e).

The ASCVD model uses the following inputs to estimate a 10-year probability of a first hard ASCVD event in adults, 40 to 79 years of age (who are free from ASCVD): age, sex, TC, HDL-C, systolic BP, use of antihypertensive therapy, diabetes, and current smoking (Goff et al. 2014). However, the USEPA (2023e) used changes in TC (in relation to PFOA and PFOS) and changes in blood pressure (BP) (in relation to PFOS, discussed further below), but not changes in high density lipoprotein cholesterol (HDL-C), as inputs to ASCVD model.

Risks of CVD are lower in individuals or populations with higher levels of HDL-C. If PFOA or PFOS are also associated with higher HDL-C, it is plausible that risks of CVD would not be impacted by higher TC (Steenland et al. 2020). In a meta-analysis, the USEPA (2023f) found that, on average, HDL-C increased with PFOA and PFOS, although the summary results of the meta-analyses were not statistically significant. Across the documents, the USEPA is not consistent in their conclusions regarding PFOA or PFOS and HDL-C. Separately, the USEPA made the following conflicting statements regarding the strength of evidence for HDL-C and PFOA:

> *"Positive associations between PFOA and HDL were also observed in most studies in the general population."* (p. 3-155, USEPA 2023a)
>
> *"HDL was not associated with PFOA."* (p. 3-173, USEPA 2023a).
>
> *"The available evidence does not support a consistent association between PFOS and reduced HDL."* (p. 3-164, USEPA 2023c).

In contrast, the USEPA presented similarly inconsistent language for effects of PFOS on BP, but included the effect of PFOS on BP in the calculation of the ASCVD model, while excluding the effect of HDL-C on PFOA or PFOS (see next section, Exposure-response function for PFOS and increases in blood pressure for additional information).

The SAB (USEPA SAB 2022a) requested that the USEPA address whether the inclusion of HDL-C would influence the results of the benefits analysis. In response, the USEPA (2023f) conducted a sensitivity analysis of a hypothetical exposure reduction of 1 ppt PFOA and 1 ppt PFOS and found that inclusion of the HDL-C effects (from the meta-analysis) decreases the annualized CVD benefits by 23-25%. Meanwhile, exclusion of the BP effects decreases annualized CVD benefits by approximately 1.8% to 2.2%. In other words, the annualized CVD benefits may be substantially overstated by excluding HDL-C from the model. In any event, the uncertainty associated with the estimated benefits from the proposed MCLs is large. It is not clear that the proposed MCLs will further drive down average PFOA or PFOS in blood serum and therefore if the estimated benefits will materialize.

4.3 Exposure-response function for PFOS and increases in blood pressure

The USEPA (2023e) also justified including changes in BP associated with PFOS as an input into the ASCVD model, and stated that the USEPA (2023c) had concluded "there was overall consistent evidence of an association between PFOS and BP in studies conducted in general adult populations." (p. 6-15, USEPA 2023e).

Subsequently, the USEPA (2023e) used the exposure-response function between PFOS and increases in blood pressure from a study of NHANES participants, 2003-2012 (Liao et al. 2020). This addition has not been peer-reviewed by the SAB and there is actually inconsistent language regarding the strength of evidence conclusions in the USEPA PFOS Toxicity Assessment (2023c),

which also indicated that the evidence of an association between PFOS and BP is uncertain. Examples of the inconsistency follow here:

> *"High and medium confidence studies reported positive associations with blood pressure and increased risk of hypertension."* (P. 3-176, USEPA 2023c)
>
> *"While there is some evidence that PFOS exposure might also have the potential to affect blood pressure and other cardiovascular responses in humans given relevant exposure circumstances, the human evidence underlying this possibility is uncertain and without support from animal or mechanistic studies."* (P. 3-176, USEPA 2023c).
>
> *"Results from studies of varying confidence reported mixed results for changes in blood pressure, including DBP and SBP, and risk of hypertension for all study populations. Studies in children (10) reported mostly non-significant associations with blood pressure and/or hypertension, though two studies in adolescents reported significantly increased (1/10) and decreased (1/10) DBP in males. In adults (13), one study reported a significantly increased risk of hypertension (1/13), but associations from other studies did not reach significance (3/13). When stratified by sex, there were mixed results. One study reported a higher risk of hypertension for males (1/13), while another reported higher risk for females (1/13). One study reported an inverse association for DBP (1/13), while others reported positive associations for DBP (6/13), but only three studies reached significance. SBP was significantly increased for all adults (4/13), in females only (2/13), and in males only (1/13). No studies examined blood pressure or hypertension in occupational populations."* [USEPA 2023c, p. 3-177]

Overall, the rationale for including changes in BP in relation to PFOS is not clear; the evidence regarding BP effects from PFOS is equivocal, similar to that of changes in HDL-C. Furthermore, to include changes in BP but not include changes in HDL-C in relation to PFOS is inconsistent, especially considering that the sensitivity analysis that included HDL-C effects in the ASCVD model showed a reduction of as much as 25% in the annualized CVD benefits if the USEPA meta-analysis slope factors are used. In contrast, exclusion of BP effects decreases annualized CVD benefits by 1.8%-2.2% if USEPA meta-analysis slope factors are used.

The estimated health benefits do not consider the potential impact of clinical management of CVD risks. That is, clinicians use the ASCVD risk model to evaluate 10-year risk of hard CVD events and inform decisions about risk management, with one of the common methods for modifying CVD risk being the use of cholesterol-lowering medications. At least two scenarios involving the use of cholesterol-lowing medications can result in overestimated CVD risk reductions in relation to PFAS based on the observed association of increased cholesterol with increased PFOA or PFOS:

1. Clinicians recommend that individuals with high cholesterol be administered cholesterol-lowering medication, with statins typically recommended first. As described in the next section on biological mechanisms, PFAS serum concentrations in individuals who use statins have not been reported to differ from serum concentrations in individuals who do not use of statins. Assuming that statins decrease circulating cholesterol levels but do not effect PFAS serum concentrations (that is, PFAS serum concentrations remain relatively unchanged), the 10-year risk of a hard CVD event will decrease due to medication use but it is unrelated to a decrease in PFAS serum concentrations. The CVD benefits calculated from reductions in PFAS levels using the ASCVD model are overstated.
2. On the other hand, the clinician may prescribe a bile acid sequestrant to lower cholesterol. As described in the next section on biological mechanisms, there is some evidence that use of bile acid sequestrants decreases PFOA serum concentrations as well as circulating cholesterol levels. In this scenario, the CVD health benefits (calculated using the ASCVD model) resulting from medication use are misattributed to decreases in PFOA (or PFOS) serum levels (whether PFAS exposure decreases or not) because the association between

PFOA and serum cholesterol is confounded by underlying physiological processes (for example, enterohepatic cycling, which could explain the reported association between bile acids and PFOA and distort the magnitude of the association between PFOA and high cholesterol).

These are theoretical examples for illustrative purposes; however, in the absence of a better understanding of the biological mechanisms that underpin the association between increases in PFOA or PFOS in blood serum in the general population and increases in total cholesterol, the quantified benefits analysis may be less than estimated.

Although the USEPA has added discussion of biological mechanisms that inform the strength of evidence conclusion for increased CVD impacts associated with PFOA and PFOS, the discussion largely focuses on biological mechanisms that inform the decreases with cholesterol seen with PFOA and PFOS at much higher serum concentrations than those reported in the general population on which the RfD is based.

Biological mechanisms for the association between PFOS or PFOA (and other PFAS) and cholesterol has not yet been identified in humans. Some information regarding potential mechanisms, however, may be gleaned from epidemiological analyses of associations between PFAS concentrations and cholesterol among those who take medications to lower cholesterol. For example:

- *Statins* (HMG-CoA reductase inhibitors) inhibit the synthesis of cholesterol in the liver and increase the removal of low-density lipoprotein cholesterol (LDL-C) that is in the blood. Andersen et al. (2021) analyzed National Health and Nutrition Examination Survey (NHANES) data from 2003 to 2016 (while accounting for NHANES sampling parameters) and reported a 2.9% increase in PFOS concentrations ($p=0.001$) among participants who reported using statins. Statin use was not associated with increased or decreased PFOA concentrations (Andersen et al. 2021). Similarly, Ma and Ducatman (2022) found that statin use was associated with statistically significantly increased PFOS concentrations and borderline significantly increased PFOA concentrations (when compared to non-users) in the C8 Health Study.
- *Bile acid sequestrants* (cholestyramine) remove bile acids that are made when LDL cholesterol breaks down. Cholestyramine lowers cholesterol by increasing bile acid secretion. Cholestyramine increased fecal elimination of PFOS and PFOA and decreased blood serum concentrations in an individual who self-administered cholestyramine (Genius et al. 2010). In the cross-sectional analysis of NHANES data from 2003-2016, Andersen et al. (2021) reported that use of cholestyramine was associated with a 1.3% reduction in PFOA and a 15.1% reduction in PFOS serum concentrations. Similarly, when compared to non-users, use of cholestyramine was associated with statistically significant decreases in serum PFAS (and the effect was strongest for PFOS) in the C8 Health Science study (Ducatman et al. 2021).
- *Probenicid* lowers cholesterol by inhibiting organic ion transporters (OAT). Probenecid helps the kidneys remove uric acid from the blood and is also used in the treatment of gout. Ducatman et al. (2021) compared Probenicid users to non-users and found that use of Probenicid was associated with a small increase in serum PFAS which was not statistically significant for PFOA or PFOS in the C8 Health Study (Ducatman et al. 2021)
- *Ezetimibe* inhibits the absorption of cholesterol in the small intestine primarily by inhibiting Niemann-Pick C1-like 1 (NPC1L1) protein. Ma and Ducatman (2022) reported that when compared to non-users, ezetimibe was not associated with blood concentrations of PFAS. Ezetimibe use was not associated with PFOA or PFOS serum concentrations in the NHANES analysis, either (Andersen et al. 2021).

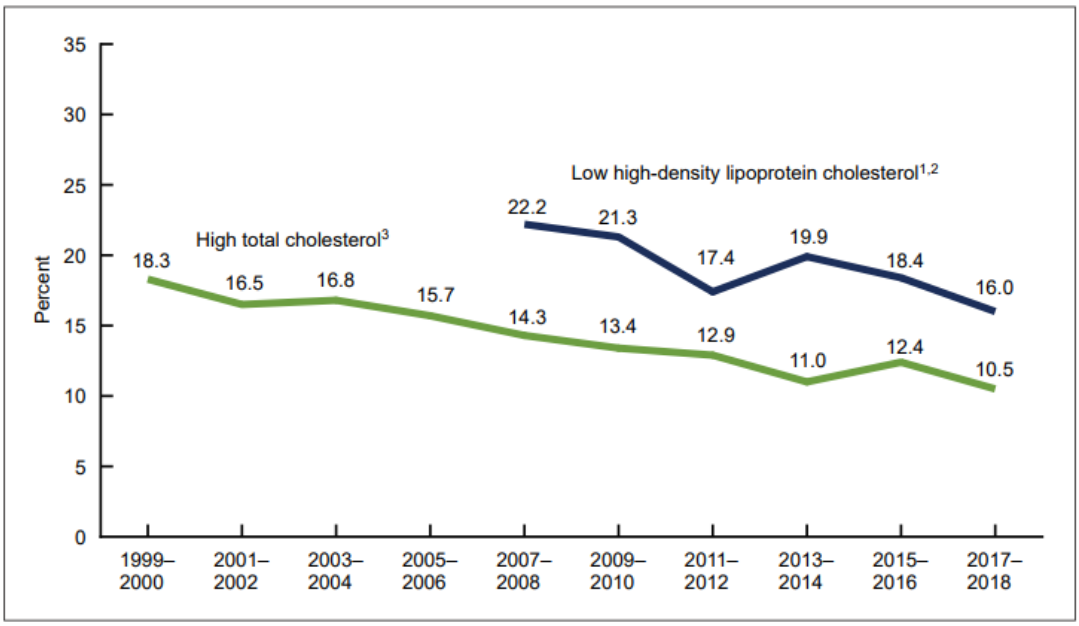
Separately, the effects of lifestyle interventions on cholesterol and PFAS blood concentrations are mixed: Morgan et al. (2023) found that lifestyle interventions over 6 months significantly reduced cholesterol; blood concentrations of PFOS and PFOA (as well as other PFAS) decreased significantly as well. After lifestyle interventions, only PFOS and total cholesterol were positively correlated and PFOS was only distributed in albumin lipoprotein fractions. Before the interventions, PFOS was found in both the albumin and non-albumin lipoprotein fractions.

4.4 Effects of other factors that mediate cholesterol levels likely dominate cardiovascular disease risks

Indirect evidence exists that suggests that the MCLs of 4.0 ppt each for PFOA and PFOS are unlikely to result in benefits as great as that reported by the USEPA (2023e) because other risk factors have a considerably larger impact on cholesterol levels. Collectively, this evidence suggests that a population shift in average cholesterol levels by further decreases in serum concentrations of PFOS and PFOA would not be detected. Over the past 20 years, PFOA and PFOS in serum have decreased in the general population. In adults 20 years and older, PFOA in serum decreased from a median of 5.20 ng/mL in 1999-2000 to 1.47 ng/mL in 2017-2018 (CDC 2023b). In adults 20 years and older, PFOS in blood serum decreased from a median of 30.3 ng/mL in 1999-2000 to 4.7 ng/mL in 2017-2018 (CDC 2023b). An examination of cholesterol levels over the past 60 years indicates substantial reductions have occurred, even during the time period when serum concentrations in PFOS and PFOA were likely increasing (before 1999-2000 when use of PFOA and PFOS in industrial applications and consumer products was greatest):

- Since 1960, TC levels have declined across all adult age groups, with the steepest declines seen in the older age groups. For example, average TC decreased from approximately 250 mg/dL in 1960 to approximately 215 mg/dL in adults 60-74 in 1999-2002 (Carroll et al. 2005). Mean TC in adults aged 20 and older declined to 188 mg/dL in 2017-2018 from 203 mg/dL in 1999-2000, a decrease of 15 mg/dL.
- In adults ages 20 and older, the prevalence of high cholesterol (total cholesterol level of at least 240 mg/dL) was 20% during 1988-1994 (Carroll et al. 2005). Since then, the prevalence of high cholesterol in adults has continued to decline from 18% during 1999-2000 to less than 11% in 2017-2018 (Carroll and Fryar 2020, Figure 4 below).
- The prevalence of high LDL-C decreased from 59% in the late 1970s to 27% in 2007-2010 (Kuklina et al. 2013). This trend was attributed to an increased percentage of adults eating diets low in saturated fats over time (from 25% during the late 1970s to 42% during 1988-1994). The percentage of adults eating diets low in saturated fats remained unchanged from 1988-1994 to 2007-2010 (Kuklina et al. 2013).
- From the late 1980s to 2007-2010, the percentage of adults using cholesterol-lowering medication increased from 5% to 23% (Kuklina et al. 2013).

Figure 4. Trends in age-adjusted prevalence of high total cholesterol and low high-density lipoprotein cholesterol among adults aged 20 and over: United States, 1999–2000 through 2017–2018



[1]Percentages prior to 2007–2008 are not presented due to changes in laboratories and methods.
[2]Significant decreasing linear trend from 2007–2008 to 2017–2018.
[3]Significant decreasing linear trend from 1999–2000 to 2017–2018.
NOTES: High total cholesterol is 240 mg/dL or more. Low high-density lipoprotein cholesterol is less than 40 mg/dL. All estimates were age adjusted by the direct method to the projected 2000 U.S. Census population using the age groups 20–39, 40–59, and 60 and over. Access data table for Figure 4 at: https://www.cdc.gov/nchs/data/databriefs/db363-tables-508.pdf#4.
SOURCE: NCHS, National Health and Nutrition Examination Survey, 1999–2018.

Source of Figure 4: Carroll and Fryar 2020.

With respect to heart disease specifically, the heart disease death rate for men aged 45-64 years in the US declined from 235.7 per 100,000 in 1999 to 183.5 per 100,000 in 2011. It increased to 192.9 per 100,000 in 2018. For women, the heart disease death rate declined from 96.8 per 100,000 in 1999 to 74.9 per 100,000 in 2011, and increased to 80.3 per 100,0000 in 2016 before leveling off (Curtin, 2020). Incidence of CVD has also declined globally over the period from 1990 to 2017 (Amini et al. 2021).

Overall, evidence of consistent decreases in heart disease incidence and mortality rates since 1990 (and earlier) suggests improvements in CVD risk factors and interventions related to diet, physical activity, and cholesterol medications are largely successful and largely drove decreases in CVD incidence and mortality until more recent years. It seems implausible that mean PFOA or PFOS serum concentrations at levels seen in the general public in recent years contribute to increased measurable risks of CVD, given that mean TC concentrations have fallen since the 1960s while PFAS blood concentrations were more likely to be increasing until the late 1990s or 2002. It is unlikely that the benefits of decreased CVD events are detectable with further declines in average blood PFAS serum concentrations.

4.5 Conclusions and Recommendations

- The USEPA should be consistent with its recent decision to include systolic blood pressure (SBP) in the ASCVD analysis related to PFOS (which is inconsistently related to SBP) and include HDL-C in the benefits analysis based on the ASCVD model. The ASCVD model uses HDL-C, and collectively, there is some evidence that PFOA and PFOS are positively correlated with HDL-C concentrations.
- The USEPA should include a more expansive discussion of biological mechanisms for the correlation of PFOA and PFOS concentrations with TC. The mechanisms which explain

decreased cholesterol with higher PFOA or PFOS serum concentrations in animals are not likely to explain the small modest increases in cholesterol in relation to small increases in PFAS concentrations in the general population.

- The USEPA should use sensitivity analyses to further explore the potential for confounding by underlying biological processes.
- The USEPA should consider whether the quantified benefits (which are substantial) make sense within the broader context of trends over time for cholesterol levels and heart disease incidence and mortality. Cholesterol levels and heart disease incidence and mortality were decreasing even before PFOA and PFOS concentrations in the blood of the general population began decreasing (since early 2000's).

5. Overall Conclusions/Comments

- The use of a HI based on different target organs or endpoints for estimation of a regulatory value has no support in existing agency guidelines or those of other national and international authoritative bodies. The agency should delay promulgating a HI-based assessment until they have developed the necessary Target Tissue Doses (TTDs), which can readily be derived using the existing ATSDR methodology (ATSDR 2018).
- Available data suggest that further reductions of average PFOA and PFOS concentrations in blood serum are unlikely to result in measurable increases in average birth weight. Data from other areas where PFOS and PFOA concentrations are elevated suggest that increases in average birth weight and decreases in infant mortality are not expected with lower PFOS and PFOA in blood serum (Olsen and Joensen 1985; Olsen et al. 2023) found that that the mean birthweight in the Faroe Islands was higher than other Nordic countries and had increased during 2010–2019 in the Faroe Islands.
- Evidence of consistent decreases in heart disease incidence and mortality rates since 1990 (and earlier) suggests improvements in CVD risk factors and interventions are largely successful. It is unlikely that mean PFOA or PFOS serum concentrations at levels seen in the general public in recent years contribute to increased measurable risks of cardiovascular disease, given that mean TC concentrations have fallen since the 1960s. It is unlikely that the benefits of decreased CVD events are detectable with further declines in average blood serum concentrations.
- Importantly, the quantified health benefits are likely to be overstated. Although epidemiological studies have reported consistent differences in biomarkers of effect (increases in total cholesterol, decreases in antibody response, increases in certain liver enzymes or small decreases in birth weight), there are only inconsistently reported increased risks of adverse health events (e.g. frequency or duration of infections) and there is generally no evidence of increased risks of low birth weight (birth weight < 2500 g) or increased risk of cardiovascular disease, the two adverse health endpoints on which the health benefits are quantified.
- Separately, it is easy to misinterpret and overstate the health benefits potentially associated with decreasing PFOA and PFOS in drinking water by "double-counting" the benefits. As evidenced by the Faroe Islands population, there is likely to be a smaller benefit than estimated (e.g. some Faroe Islands birth cohorts showed decreases in anti-diphtheria and anti-tetanus responses in relation to PFOA and PFOS in birth cohorts that have some of the largest mean birth weights globally). In other words, any quantifiable health benefit for the population would apply to some combination of endpoints (some small benefits associated with increased birth weight, some small benefits associated with decreased cholesterol) but not the cumulative endpoints (benefits associated with decreased birth weight plus benefits associated with decreased cardiovascular diseases).

Because the actual nature of the mechanisms for the observed associations between PFOA and PFOS and the health effects are unknown, there remains substantial uncertainty in the range of the quantified benefits associated with the Proposed Option as well as the alternative regulatory options.

6. References

Amini M, Zayeri F, Salehi M. 2021. Trend analysis of cardiovascular disease mortality, incidence, and mortality-to-incidence ratio: results from global burden of disease study 2017. BMC Public Health, 21: 401. doi: 10.1186/s12889-021-10429-0.

Andersen ME, Hagenbuch B, Apte U, Corton JC, Fletcher T, Lau C, Roth WL, Staels B, Vega GL, Clewell HJ, Longnecker MP. 2021. Why is elevation of serum cholesterol associated with exposure to perfluoroalkyl substances (PFAS) in humans? A workshop report on potential mechanisms. Toxicology, 459: 152845. doi:10.1016/j.tox.2021.152845.

ATSDR. 2018. Framework for Assessing Health Impacts of Multiple Chemicals and Other Stressors (Update). Agency for Toxic Substances and Disease Registry. Available at: https://www.atsdr.cdc.gov/interactionprofiles/ipga.html.

ATSDR. 2021. Toxicological Profile for Perfluoroalkyls. Agency for Toxic Substances and Disease Registry. Available at: https://www.atsdr.cdc.gov/toxprofiles/tp200.pdf.

ATSDR. 2022. Public Health Assessment Guidance Manual (PHAGM). Agency for Toxic Substances and Disease Registry. Available at: https://www.atsdr.cdc.gov/pha-guidance/index.html.

Behrman JR, Rosenzweig MR. 2004. Returns to birthweight. Review of Economics and Statistics, 86(2): 586-601.

Behrman RE, Butler AS. 2007. Preterm birth: causes, consequences, and prevention. Institute of Medicine of the National Academies.

Carroll MD, Lacher DA, Sorlie PD, Cleeman JI, Gordon DJ, Wolz M, Grundy SM, Johnson CL. 2005. Trends in serum lipids and lipoproteins of adults, 1960--2002. JAMA, 294: 1773-1781. doi: 10.1001/jama.294.14.1773.

Carroll MD, Fryar CD. 2020. Total and High-density Lipoprotein Cholesterol in Adults: United States, 2015–2018. NCHS data brief, no. 363. Hyattsville, Maryland: National Center for Health Statistics. April 2020.

CDC. 2023a. Natality Data for 1995-2021. Centers for Disease Control and Prevention, United States Department of Health and Human Services. Available at: https://wonder.cdc.gov/natality.html.

CDC. 2023b. National Report on Human Exposure to Environmental Chemicals. Centers for Disease Control and Prevention, United States Department of Health and Human Services. Updated March 2022. Accessed April 6, 2023. https://www.cdc.gov/exposurereport/.

Chaikind S, Corman H. 1991. The impact of low birthweight on special education costs. Journal of Health Economics, 10(3): 291-311. doi: 10.1016/0167-6296(91)90031-h.

Château-Degat ML, Pereg D, Dallaire R, Ayotte P, Dery S, Dewailly É. 2010. Effects of perfluorooctanesulfonate exposure on plasma lipid levels in the Inuit population of Nunavik (Northern Quebec). Environ Res 110(7): 710-717. doi: 10.1016/j.envres.2010.07.003.

Chatterji P, Kim D, Lahiri K. 2014. Birth weight and academic achievement in childhood. Health Economics, 23(9): 1013-1035. doi: 10.1002/hec.3074.

Chen MH, Ha EH, Wen TW, Su YN, Lien GW, Chen CY, Chen PC, Hsieh WS. 2012. Perfluorinated compounds in umbilical cord blood and adverse birth outcomes. PLoS ONE, 7(8): e42474. doi: 10.1371/journal.pone.0042474.

Chu C, Zhou Y, Li QQ, Bloom MS, Lin S, Yu YJ, Chen D, Yu HY, Hu LW, Yang BY, Zeng XW, Dong GH. 2020. Are perfluorooctane sulfonate alternatives safer? N ew insights from a birth cohort study. Environment International, 135: 105365. doi: 10.1016/j.envint.2019.105365.

Colaizy TT, Bartick MC, Jegier BJ, Green BD, Reinhold AG, Schaefer AJ, Bogen DL, Schwarz EB, Stuebe AM; Eunice Kennedy Shriver National Institute of Child Health and Human Development Neonatal Research Network. 2016. Impact of optimized breastfeeding on the costs of necrotizing enterocolitis in extremely low birthweight infants. The Journal of Pediatrics, 175: 100-105. e102.

Curtin SC. 2020. Cancer and Heart Disease Death Rates, Among Men and Women Aged 45–64 Years — United States, 1999–2018. MMWR Morbidity and Mortality Weekly Report, 69: 658. doi: 10.15585/mmwr.mm6921a4.

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. 2013 121(10):1207-13. doi: 10.1289/ehp.1206372.

Dobson KG, Ferro MA, Boyle MH, Schmidt LA, Saigal S, Van Lieshout RJ. 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. doi: 10.1017/S0954579417001651.

Dong Z, Wang H, Yu YY, Li YB, Naidu R, 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. doi: 10.1016/j.ecoenv.2019.02.061.

Ducatman A, Luster M, Fletcher T. 2021. Perfluoroalkyl substance excretion: Effects of organic anion-inhibiting and resin-binding drugs in a community setting. Environ Toxicol Pharmacol 85:103650. doi: 10.1016/j.etap.2021.103650.

Dzierlenga M, Crawford L, Longnecker M. 2020. Birth weight and perfluorooctane sulfonic acid: a random-effects meta-regression analysis. Environmental Epidemiology, 4: e095. doi: 10.1097/EE9.0000000000000095.

Elder T, Figlio D, Imberman S, Persico C. 2020. The role of neonatal health in the incidence of childhood disability. American Journal of Health Economics, 6(2): 216-250.

EFSA (European Food Safety Authority), Knutsen HK, Alexander J, Barregard L, Bignami M, Bruschweiler B, Ceccatelli S, Cottrill B, Dinovi M, Edler L, Grasl-Kraupp B, Hogstrand C, Hoogenboom LR, Nebbia CS, Oswald IP, Petersen A, Rose M, Roudot AC, Vleminckx C, Vollmer G, Wallace H, Bodin L, Cravedi JP, Halldorsson TI, Haug LS, Johansson N, van Loveren H, Gergelova P, Mackay K, Levorato S, van Manen M, 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.

EFSA (European Food Safety Authority), Schrenk D, Bignami M, Bodin L, Chipman JK, Del Mazo J, Grasl-Kraupp B, Hogstrand C, Hoogenboom LR, Leblanc JC, Nebbia CS, Nielsen E, Ntzani E, Petersen A, Sand S, Vleminckx C, Wallace H, Barregard L, Ceccatelli S, Cravedi JP, Halldorsson TI, Haug LS, Johansson N, Knutsen HK, Rose M, Roudot AC, Van Loveren H, Vollmer G, Mackay K, Riolo F, Schwerdtle T. 2020. Risk to human health related to the presence of perfluoroalkyl substances in food. EFSA Journal, 18(9): e06223.

Fan Y, Li X, Xu Q, Zhang Y, Yang X, Han X, Du G, Xia Y, Wang X, Lu C. 2020. Serum albumin mediates the effect of multiple per-and polyfluoroalkyl substances on serum lipid levels. Environmental Pollution, 266: 115138. doi: 10.1016/j.envpol.2020.115138.

Fei C, McLaughlin JK, Tarone RE, Olsen J. 2007. Perfluorinated chemicals and fetal growth: A study within the Danish National Birth Cohort. Environmental Health Perspectives, 115: 1677-1682. doi: 10.1289/ehp.10506.

Fei C, McLaughlin JK, Tarone RE, Olsen. 2008. Fetal growth indicators and perfluorinated chemicals: A study in the Danish National Birth Cohort. American Journal of Epidemiology, 168(1): 66-72. doi: 10.1093/aje/kwn095.

Genuis SJ, Birkholz D, Ralitsch M, Thibault N. 2010. Human detoxification of perfluorinated compounds. Public Health, 124: 367–375. doi:10.1016/j.puhe.2010.03.002.

Goff DC, Lloyd-Jones DM, Bennett G, Coady S, D'Agostino RB, Gibbons R, Greenland P, Lackland DT, Levy D, O'Donnell CJ, Robinson JG, Schwartz JS, Shero ST, Smith SC, Sorlie P, Stone NJ, Wilson PW. 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: S49-S73. doi: 10.1161/01.cir.0000437741.48606.98.

Govarts E, Remy S, Bruckers L, Den Hond E, Sioen I, Nelen V, Baeyens W, Nawrot TS, Loots I, Van Larebeke N, Schoeters G. 2016. Combined Effects of Prenatal Exposures to Environmental Chemicals on Birth Weight. International Journal of Environmental Research and Public Health, 13: 1-19. doi:10.3390/ijerph13050495.

Grandjean P, Andersen EW, Budtz-Jørgensen E, Nielsen F, Mølbak K, Weihe P, Heilmann C. 2012. Serum vaccine antibody concentrations in children exposed to perfluorinated compounds. JAMA, 307(4): 391-397.

Hamm MP, Cherry NM, Chan E, Martin JW, Burstyn I. 2010. Maternal exposure to perfluorinated acids and fetal growth. Journal of Exposure Science and Environmental Epidemiology, 20(7):589-597. doi: 10.1038/jes.2009.57.

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. The Science of the Total Environment, 625: 566-574. doi: 10.1016/j.scitotenv.2017.12.186.

Hjermitslev MH, Long M, Wielsøe M, Bonefeld-Jørgensen EC. 2020. Persistent organic pollutants in Greenlandic pregnant women and indices of foetal growth: The ACCEPT study. The Science of the Total Environment, 698: 134118. doi: 10.1016/j.scitotenv.2019.134118.

Hines CT, Padilla CM, Ryan RM. 2020. The effect of birth weight on child development prior to school entry. Child development, 91(3): 724-732. doi: 10.1111/cdev.13355.

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 MA, Fagnani C, Brescianini S, Derom CA, Vlietinck RF, Loos RJF, Krueger RF, McGue M, Pahlen S, Nelson TL, Whitfield KE, Brandt I, Nilsen TS, Harris JR, Cutler TL, Hopper JL, Tarnoki AD, Tarnoki DL, Sørensen TIA, Kaprio J, Silventoinen K. 2018. Association between birth weight and educational attainment: an individual-based pooled analysis of nine twin cohorts. Journal of Epidemiology and Community Health, 72(9): 832-837. doi: 10.1136/jech-2017-210403.

Joyce C, Goodman-Bryan M, Hardin A. 2012. Preterm Birth and Low Birth Weight. Retrieved from The Urban Child Institute: http://www.urbanchildinstitute.org/sites/all/files/2010-10-01-PTB-and-LBW.pdf.

Klein R, Lynch M. 2018. Development of Medical Cost Estimates for Adverse Birth Outcomes. Prepared for U.S. EPA National Center for Environmental Economics.

Kowlessar NM, Jiang HJ, Steiner C. 2013. Hospital stays for newborns, 2011: statistical brief# 163.

Kuklina EV, Carroll MD, Shaw KM, Hirsch R. 2013. Trends in High LDL Cholesterol, Cholesterol-lowering Medication Use, and Dietary Saturated-fat Intake: United States, 1976–2010. NCHS data brief, No. 117. Hyattsville, MD: National Center for Health Statistics. March 2013.

Lauritzen HB, Larose TL, Øien T, Sandanger TM, Odland JØ, van de Bor M, Jacobsen GW. 2018. Prenatal exposure to persistent organic pollutants and child overweight/obesity at 5-year follow-up: a prospective cohort study. Environmental Health, 17: 9. doi: 10.1186/s12940-017-0338-x.

Liao S, Yao W, Cheang I, Tang X, Yin T, Lu X, Zhou Y, Zhang H, Li X. 2020. Association between perfluoroalkyl acids and the prevalence of hypertension among US adults. Ecotoxicology and Environmental Safety, 196: 110589. doi: 10.1016/j.ecoenv.2020.110589.

Ma G, Ducatman A. 2022. Perfluoroalkyl Substance Serum Concentrations and Cholesterol Absorption-Inhibiting Medication Ezetimibe. Toxics, 10: 799. doi: 10.3390/toxics10120799.

Manzano-Salgado CB, Casas M, Lopez-Espinosa MJ, Ballester F, Iñiguez C, Martinez D, Romaguera D, Fernández-Barrés S, Santa-Marina L, Basterretxea M, Schettgen T, Valvi D, Vioque J, Sunyer J, Vrijheid M. 2017. Prenatal exposure to perfluoroalkyl substances and cardiometabolic risk in children from the Spanish INMA birth cohort study. Environmental Health Perspectives, 125: 097018. doi: 10.1186/s12940-017-0338-x.

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. International Journal of Environmental Research and Public Health, 15(9): 1832. doi: 10.3390/ijerph15091832.

Morgan S, Mottaleb MA, Kraemer MP, Moser DK, Worley J, Morris AJ, Petriello MC. 2023. Effect of lifestyle-based lipid lowering interventions on the relationship between circulating levels of per- and polyfluoroalkyl substances and serum cholesterol. Environmental Toxicology and Pharmacology, 98: 104062. doi: 10.1016/j.etap.2023.104062.

Mumtaz MM, Poirier KA, Coleman JT. 1997. Risk assessment for chemical mixtures: fine tuning the hazard index approach. Journal of Clean Technology Environmental Toxicology and Occupational Medicine, 6(2): 189-204.

Negri E, Metruccio F, Guercio V, Tosti L, Benfenati E, Bonzi R, La Vecchia C, Moretto A. 2017. Exposure to PFOA and PFOS and fetal growth: a critical merging of toxicological and epidemiological data. Critical Reviews in Toxicology, 47: 482-508. doi: 10.1080/10408444.2016.1271972.

Nelson JW, Hatch EE, Webster TF. 2010. Exposure to polyfluoroalkyl chemicals and cholesterol, body weight, and insulin resistance in the general US population. Environmental Health Perspectives, 118(2): 197-202. doi: 10.1289/ehp.0901165.

Nicoletti C, Salvanes KG, Tominey E. 2018. Response of parental investments to child's health endowment at birth. In Health Econometrics: Emerald Publishing Limited.

Nolan LA, Nolan JM, Shofer FS, Rodway NV, Emmett EA. 2009. The relationship between birth weight, gestational age and perfluorooctanoic acid (PFOA)-contaminated public drinking water. Reproductive Toxicology, 27: 231-238. doi: 10.1016/j.reprotox.2008.11.001.

Olsen SF, Joensen HD. 1985. High liveborn birth weights in the Faroes: a comparison between birth weights in the Faroes and in Denmark. Journal of Epidemiology and Community Health, 39: 27-32. doi: 10.1136/jech.39.1.27.

Olsen SH, Reynstind D, Hallgrímsson H, Kesmodel US. 2023. Birthweight and gestational age in the Faroe Islands: A comparison between birthweight and gestational age in the Faroe Islands and other Nordic countries. Acta Obstetricia et Gynecologica Scandinavica, 102: 506-515. doi: 10.1111/aogs.14527.

Sagiv SK, Rifas-Shiman SL, Fleisch AF, Webster TF, Calafat AM, Ye X, Gillman MW, Oken E. 2018. Early-pregnancy plasma concentrations of perfluoroalkyl substances and birth outcomes in Project Viva: confounded by pregnancy hemodynamics? American Journal of Epidemiology, 187: 793-802. doi: 10.1093/aje/kwx332.

Savitz DA, Stein CR, Bartell SM, Elston B, Gong J, Shin HM, Wellenius GA. 2012a. Perfluorooctanoic acid exposure and pregnancy outcome in a highly exposed community. Epidemiology, 23(3): 386-392. doi: 10.1097/EDE.0b013e31824cb93b.

Savitz DA, Stein CR, Elston B, Wellenius GA, Bartell SM, Shin HM, Vieira VM, Fletcher T. 2012b. Relationship of perfluorooctanoic acid exposure to pregnancy outcome based on birth records in the Mid-Ohio Valley. Environmental Health Perspectives, 120(8): 1201-1207. doi: 10.1289/ehp.1104752.

Starling AP, Adgate JL, Hamman RF, Kechris K, Calafat AM, Ye X, 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. Environmental Health Perspectives, 125: 067016-067011. doi:10.1289/EHP641.

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. American Journal of Epidemiology, 170(10): 1268-1278. doi: 10.1093/aje/kwp279.

Steenland K, Barry V, Savitz D. 2018. Serum Perfluorooctanoic Acid and Birthweight an Updated Meta-analysis With Bias Analysis. Epidemiology, 29: 765–776. doi: 10.1097/EDE.0000000000000903.

Steenland K, Fletcher T, Stein CR, Bartell SM, Darrow L, Lopez-Espinosa MJ, Ryan PB, Savitz DA. 2020. Review: Evolution of evidence on PFOA and health following the assessments of the C8 Science Panel. Environment International, 145: 106125. doi: 10.1016/j.envint.2020.106125.

Stein CR, Savitz DA, Dougan M. 2009. Serum levels of perfluorooctanoic acid and perfluorooctane sulfonate and pregnancy outcome. American Journal of Epidemiology, 170(7): 837-846. doi: 10.1093/aje/kwp212.

Temple JA, Reynolds AJ, Arteaga I. 2010. Low birth weight, preschool education, and school remediation. Education and urban society, 42(6): 705-729. doi: 10.1177/0013124510370946.

Tilstra AM, Masters RK. 2020. Worth the Weight? Recent Trends in Obstetric Practices, Gestational Age, and Birth Weight in the United States. Demography, 57:99-121. doi: 10.1007/s13524-019-00843-w.

USEPA. 1986. Guidelines for health risk assessment of chemical mixtures. United States Environmental Protection Agency. Federal Register 51(185): 34014-34025.

USEPA. 2000. Supplementary Guidance for Conducting Health Risk Assessment of Chemical Mixtures. United States Environmental Protection Agency. Risk Assessment Forum Technical Panel. Washington, DC. EPA/630/R-00/002.

USEPA. 2016a. Health Effects Support Document for Perfluorooctanoic Acid (PFOA). United States Environmental Proection Agency, Office of Water. EPA Document No: 822-R-16-003. May 2016.

USEPA. 2016b. Health Effects Support Document for Perfluorooctane Sulfonate (PFOS). United States Environmental Proection Agency, Office of Water. EPA Document No: 822-R-16-002. May 2016.

USEPA. 2021a. External Peer Review Draft. Proposed Approaches to the Derivation of a Draft Maximum Contaminant Level Goal for Perfluorooctanoic Acid (PFOA) (CASRN 335-67-1) in Drinking Water. United States Environmental Protection Agency, Office of Water. EPA Document No: 822D21001. November 2021.

USEPA. 2021b. External Peer Review Draft. Proposed Approaches to the Derivation of a Draft Maximum Contaminant Level Goal for Perfluorooctane Sulfonic Acid (PFOS) (CASRN 1763-23-1) in Drinking Water. United States Environmental Protection Agency, Office of Water. EPA Document No: 822D21001. November 2021.

USEPA. 2021c. External Peer Review Draft. Draft Framework for Estimating Noncancer Health Risks Associated with Mixtures of Per- and Polyfluoroalkyl Substances (PFAS). United States Environmental Protection Agency, Office of Water. EPA Document No: 822D21001. November 2021.

USEPA. 2021d. Analysis of Cardiovascular Disease Risk Reduction as a Result of Reduced PFOA and PFOS Exposure in Drinking Water. United States Environmental Protection Agency, Office of Water. EPA Document No: 815-D-21-001. November 2021.

USEPA. 2022a. Transmittal of the Science Advisory Board Report titled, "Review of EPA's Analyses to Support EPA's National Primary Drinking Water Rulemaking for PFAS." Science Advisory Board – A Federal Advisory Committee to the United States Environmental Protection Agency. August 22, 2022. EPA-SAB-22-008. Available at: https://sab.epa.gov/ords/sab/f?p=100:18:16490947993:::RP,18:P18_ID:2601.

USEPA. 2022b. America's Children and the Environment (ACE), Third Edition. Biomonitoring – Perfluorochemicals (PFCs). United States Environmental Protection Agency. Accessed April 25, 2023. https://www.epa.gov/americaschildrenenvironment/biomonitoring-perfluorochemicals-pfcs.

USEPA. 2022c. EPA Office of Research and Development Integrated Risk Information System (IRIS) Program Outlook. United States Environmental Protection Agency. June 2022. https://www.epa.gov/system/files/documents/2022-06/IRIS%20Program%20Outlook_June22.pdf.

USEPA. 2023a. Public Comment Draft Toxicity assessment and proposed maximum contaminant level goal for perfluorooctanoic acid (PFOA) in drinking water. United States Environmental Protection Agency. EPA Document No. 822P23005.

USEPA. 2023b. Public Comment Draft Appendix: Toxicity assessment and proposed maximum contaminant level goal for perfluorooctanoic acid (PFOA) in Drinking Water. United States Environmental Protection Agency. EPA Document No. 822P23006.

USEPA. 2023c. Public Comment Draft Toxicity assessment and proposed maximum contaminant level goal for perfluorooctane sulfonic acid (PFOS) in drinking water. United States Environmental Protection Agency. EPA Document No. 822P23007.

USEPA. 2023d. Public Comment Draft Appendix: Toxicity assessment and proposed maximum contaminant level goal for perfluorooctanoic sulfonic acid (PFOS) in Drinking Water. United States Environmental Protection Agency. EPA Document No. 822P23008.

USEPA. 2023e. Economic analysis for the proposed per- and polyfluoroalkyl substances national primary drinking water regulation. United States Environmental Protection Agency. EPA-822-P-23-001.

USEPA. 2023f. Economic analysis for the proposed per- and polyfluoroalkyl substances national primary drinking water regulation appendices. United States Environmental Protection Agency. EPA-822-P-23-002.

USEPA. 2023g. Public Review Draft. Maximum contaminant level goal (MCLG) summary document for a mixture of four per- and polyfluoroalkyl substances (PFAS): HFPO-DA and its ammonium salt (also known as GenX chemicals), PFBS, PFNA and PFHxS. United States Environmental Protection Agency. EPA-822-P-23-004.

USEPA. 2023h. Public Review Draft. Framework for estimating noncancer health risks associated with mixtures of per- and polyfluoroalkyl substances (PFAS). United States Environmental Protection Agency. EPA-822-P-23-003.

USEPA. 2023i. 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. United States Environmental Protection Agency. Washington, DC 20460. EPA Document No. 815D23001.

Verner MA, Loccisano AE, Morken NH, Yoon M, Wu H, McDougall R, Maisonet M, Marcus M, Kishi R, Miyashita C, Chen MH, Hsieh WS, Andersen ME, Clewell HJ 3rd, Longnecker MP. 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. doi:10.1289/ehp.1408837.

Wang Y, Miao Y, Mir AZ, Cheng L, Wang L, Zhao L, Cui Q, Zhao W, Wang H. 2016. Inhibition of beta-amyloid-induced neurotoxicity by pinocembrin through Nrf2/HO-1 pathway in SH-SY5Y cells. Journal of the Neurological Sciences, 368: 223-230. doi: 10.1016/j.jns.2016.07.010.

Whitworth KW, Haug LS, Baird DD, Becher G, Hoppin JA, Skjaerven R, Thomsen C, Eggesbo M, Travlos G, Wilson R, Cupul-Uicab LA, Brantsaeter AL, Longnecker MP. 2012. Perfluorinated compounds in relation to birth weight in the Norwegian Mother and Child Cohort Study. American Journal of Epidemiology, 175(12): 1209-1216. doi: 10.1093/aje/kwr459.

Wikström S, Lin PI, Lindh CH, Shu H, Bornehag CG. 2020. Maternal serum levels of perfluoroalkyl substances in early pregnancy and offspring birth weight. Pediatric Research, 87: 1093-1099. doi: 10.1038/s41390-019-0720-1.

Wu K, Xu X, Peng L, Liu J, Guo Y, Huo X. 2012. Association between maternal exposure to perfluorooctanoic acid (PFOA) from electronic waste recycling and neonatal outcomes. Environment International, 48: 1-8. doi: 10.1016/j.envint.2012.06.018.

Yao Q, Gao Y, Zhang Y, Qin K, Liew Z, 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. doi: 10.1016/j.chemosphere.2021.131521.

American Water Works Association
Dedicated to the World's Most Important Resource™

Government Affairs Office
1300 Eye Street NW
Suite 701W
Washington, DC 20005-3314
T 202.628.8303
F 202.628.2846

December 30, 2021

Dr. Weihsueh A. Chiu, Ph.D.
Chair
Science Advisory Board PFAS Review Panel
Environmental Protection Agency
1300 Pennsylvania Ave NW
Washington, DC 20460

SUBMITTED ELECTRONICALLY

RE: AWWA Comments for Science Advisory Board PFAS Review Panel Consideration

Dear Dr. Chiu,

The American Water Works Association (AWWA) appreciates the public service provided by the Science Advisory Board (SAB) PFAS Review Panel members. The documents the Panel is reviewing will lay the foundation for the national primary drinking water regulation (NPDWR) for at least two per- and polyfluoroalkyl substances (PFAS) in drinking water and perhaps other PFAS through EPA's current effort and future rulemakings. AWWA looks forward to the SAB Panel's feedback. As the Panel is aware, the Safe Drinking Water Act (SDWA) and Executive Orders set a clear bar for transparent analysis and use of sound science in setting NPDWRs. AWWA appreciates the Panel's assistance to EPA in this process. AWWA offers the following comments for the Panel's consideration in its deliberation and for the Agency as it addresses the Panel's review.

Sufficient Resources are Needed to Ensure Scientific Integrity

Under the SDWA, the EPA has a responsibility to use the best-available science in accordance with sound and objective scientific practices. Doing so is imperative in ensuring that drinking water contaminants are addressed in a meaningful way that protects the public and can be implemented feasibly. As EPA has already presented and discussed during the Panel's Dec. 16 meeting, there are five significant draft documents for your consideration relating to the draft approaches for developing a perfluorooctanoic acid (PFOA) MCLG and a perfluorooctanesulfonic acid (PFOS) MCLG, evaluating cardiovascular health effects of PFAS, and evaluating non-cancer health risks associated with PFAS mixtures. This represents a substantial body of work for the Panel to review and on which to provide feedback.

The SAB PFAS Review Panel's report of recommendations is anticipated to be completed in May 2022. If EPA adheres to the schedule set in its PFAS Strategic Roadmap, this leaves less than six months for the Agency to review these recommendations, implement appropriate changes, incorporate resulting differences into its cost analysis, craft a regulatory proposal, and complete the associated procedural

requirements for proposing rulemakings. Normally one-third of that six-month period would be taken up by inter-Agency review. This schedule is well within EPA's statutory deadline for proposal of the rule. AWWA urges the EPA Administrator to afford the SDWA program staff sufficient resources to act on your review so that the proposal is based on the best available science and that the analyses are sound. This is a critical rulemaking for the program, and the Administrator must be sure that EPA employs a defensible premise for benefits anticipated to be accrued through this rulemaking.

Charge Questions and Ensuring Recommendations Reflect Purpose of Documents

On Dec. 16, EPA presented the Panel with its intended use of the documents being reviewed. The Agency made clear why it included five economists in a Panel charged with reviewing scientific assessments of health data. Unfortunately, the final charge questions distributed to the Panel do not address the key questions EPA must answer or make best use of the expertise of the economists on the Panel.

To inform its deliberations, AWWA recommends that the Panel request a briefing from the National Center for Economic Analysis on the construction of benefits analyses to support SDWA regulatory standard setting. Such a briefing would provide the Panel members a common understanding of the task before EPA using the materials the Panel is reviewing. With that basis, the panelists with expertise in national benefit analysis would be able to speak to the strengths and weaknesses of the Agency approach for purposes of the benefits analysis. For example, the Agency is positing in one of its analyses that a fraction of cardiovascular disease in the United States is attributable to PFOA and PFOS. The U.S. Centers for Disease Prevention and Control currently estimates that the mortality rate for diseases of the heart in the United States is 200.8 per 100,000. There are recognized risk factors, some of which have marked socio-economic correlations. It is important that EPA (1) neither grossly over- or under-estimate benefits from risk reduction in its rulemakings and (2) understand and communicate how assumptions and uncertainties in its analysis impact use of the risk reduction model. The plausibility of the Agency's analytical approach being adequate to underpin an economic analysis for an SDWA primary standard warrants discussion.

In-Depth Review of the Documents Provided for Panel Review

AWWA contracted with Ramboll U.S. Consulting, Inc. (Ramboll) to prepare a review of the documents before the Panel and compile comments relative to the charge questions posed to the Panel. The scientists at Ramboll included experts in both cancer and non-cancer health risk assessments, physiologically based pharmacokinetic modeling, and epidemiological research. A summary of the review by Ramboll scientists is attached and is organized to align with the Agency's charge to the Panel. Some key points are:

1. EPA did not apply the Agency's current systematic review process to all the studies it utilized; studies central to its quantitative analysis were likely not held to the expectations applied in the current systematic review process.

2. The evidence for a causal relationship between PFOA or PFOS exposure and cardiovascular disease is weak. It is plausible that PFOA and PFOS exposure is associated with higher cholesterol levels but without an increased risk of cardiovascular disease. Because epidemiological evidence of increased risks of cardiovascular disease in relation

to PFOA and PFOS exposure is weak, it is currently speculative to assume that the small increases in total cholesterol or low-density lipoprotein cholesterol (LDL-C, the "bad" cholesterol) are causally related to increased incidence of cardiovascular disease.

- EPA utilizes studies based on an apparent association between PFOA and PFOS exposure and total cholesterol/ LDL-C when the weight-of-evidence is limited and there are scientific reasons to suspect the association is not meaningful for EPA's analysis.
- Furthermore, if PFOA and PFOS are associated with small increases in high density lipoprotein cholesterol (HDL-C), it is biologically plausible that the risks of cardiovascular disease remain unchanged. HDL-C is considered to be the "good" cholesterol and higher levels of HDL-C are associated with a decreased risk of cardiovascular disease. The EPA should consider this as part of the analysis.
- The analysis presented in the cardiovascular disease risk reduction document concludes that PFOA and PFOS levels lead to an increase in cholesterol. This is not a certainty, and it is possible that the correlation between PFOA and PFOS levels and cholesterol levels is more likely related to the transport mechanisms of cholesterol and PFOA and PFOS within the body.

3. Shearer et al. (2021), one of the key studies in EPA's analyses, should not be used as a basis for either cancer characterization or dose-response assessment. Despite analyses that adjusted for estimated glomerular filtration, there is the possibility of additional confounding by effects of the underlying cancer induction processes on other aspects of kidney function, such as the renal transporters that are required for control of PFOA excretion, that could also lead to higher PFOA blood concentrations. In addition, the maximum latency in the study was 18 years since the blood collection, which is generally inadequate for kidney cancer, which has a long disease latency. Separately, there is an apparent anomaly between the number of cases and controls in the referent category of exposure (<4 ng/mL) which may lead to under- or over-estimated odds ratios in the higher quartiles of exposure.

4. EPA's analyses for PFOA and PFOS rely upon a half-life from a study of retired workers exposed occupationally (Olsen et al., 2007) and thought to have been exposed intermittently since retirement. Recent work by the Alliance for Risk Assessment concluded that the most appropriate studies support a much lower half-life. EPA acknowledges the study's shortcomings but does not provide reasoning for nonetheless using the higher half-lives.

5. EPA's analyses rely on human epidemiological studies with published findings of reduced vaccination efficacy based on cohorts in the Faroe Islands. There are several concerns associated with these studies relating to the level of clinical protection, inconsistencies regarding study subjects included in various studies, and confounding resulting from contaminants (e.g., polychlorinated biphenyl, methyl mercury) anticipated to be high in

the Faroese diet. Should the EPA decide to use these studies, the data should be obtained from the study investigators and independently evaluated prior to finalizing the Agency's analysis.

6. Several key aspects of these analyses are not presented for the SAB PFAS Review Panel and the public to review and verify, including:

    - Neither EPA's animal nor human pharmacokinetic model files were made available for public verification. Without these model files, the model cannot be verified. Lack of access to files also limits the ability to evaluate the pregnancy and lactation model. All model files (including R scripts) should be made available for review by the SAB PFAS Review Panel, as well as the public, to provide scientific transparency.

    - The source for the milk ingestion for animal pups was not fully documented, nor was it peer reviewed. EPA should fully describe the basis for milk ingestion in the PFOA and PFOS documents to adequately support the Panel's review.

7. The draft framework for assessing non-cancer health effects of PFAS mixtures is a significant improvement upon previous approaches applied by other regulatory agencies. However, the document relies on the conclusion that dose additivity occurs for chemicals with a similar toxic endpoint. This begs the question of "How similar?". While chemicals may share a common toxicity endpoint, if the health effects vary, then it becomes necessary to define sub-classes of chemicals for which dose-additivity is appropriate.

If you have any questions regarding this correspondence or if we can be of assistance in some other way, please contact Chris Moody (202.326.6127, cmoody@awwa.org).

Best regards,

FOR THE AMERICAN WATER WORKS ASSOCIATION

G. Tracy Mehan, III
Executive Director – Government Affairs
American Water Works Association

cc: Suhair Shallal Al-Mudallal, EPA/SAB
Al McGartland, EPA/OP/NCEE
Erik Helm, EPA/OW
Betsy Behl, EPA/OW/OST
Jennifer McLain, EPA/OW/OGWDW
Eric Burneson, EPA/OW/OGWDW/SRMD

Attachment (1)

Who is AWWA

The American Water Works Association (AWWA) is an international, nonprofit, scientific and educational society dedicated to providing total water solutions assuring the effective management of water. Founded in 1881, the Association is the largest organization of water supply professionals in the world. Our membership includes more than 4,500 utilities that supply roughly 80 percent of the nation's drinking water and treat almost half of the nation's wastewater. Our 50,000-plus total membership represents the full spectrum of the water community: public water and wastewater systems, environmental advocates, scientists, academicians, and others who hold a genuine interest in water, our most important resource. AWWA unites the diverse water community to advance public health, safety, the economy, and the environment.

ATTACHMENT 1

Technical Expert Review of

Agency Review Documents Relating to PFAS Health Risks in Drinking Water

Technical Expert Review of
Agency Review Documents Relating to PFAS Health Risks in Drinking Water

Prepared by
Ramboll US Consulting, Inc.
3107 Armand Street
Monroe, LA 71201

Prepared for:
American Water Works Association
1300 Eye Street NW
Suite 701W
Washington, DC 20005-3314

December 23, 2021

TABLE OF CONTENTS

PROPOSED APPROACHES TO THE DERIVATION OF A DRAFT MAXIMUM CONTAMINANT LEVEL GOALS FOR PFOA AND PFOS IN DRINKING WATER

General Comment: *As noted in the background sections for both documents, because PFOA and PFOS are listed on the Fourth Drinking Water Contaminant Candidate List (USEPA 2021), EPA made a determination to regulate PFOA and PFOS with a National Primary Drinking Water Regulation. While there are separate documents to discuss the approach for the development of the Maximum Contaminant Level Goal (MCLG) for each compound, there is a lot of information focused on the combination of these compounds or additional PFAS as a class of compounds. For example, in the Occurrence Summary (Section 1.4), examination of the occurrence relies upon the data from the third Unregulated Contaminant Monitoring Rule (from 2013-2015; Section 1.4) and is focused on the occurrence of PFOS and PFOA in aggregate, summing concentrations. This concept, as well as others throughout the documents, should be focused on each compound separately when considering data in the estimation of the individual MCLGs for each compound.*

Study Identification and Inclusion

1. EPA used systematic review methods consistent with the current ORD systematic review practice to ensure transparency and completeness of literature identification, sorting, and study quality evaluation. Is the process clearly described? Please identify additional peer-reviewed studies that the panel is aware of that could inform toxicity value derivation.

Comment: *As noted in both the PFOA and PFOS documents, EPA has built upon the data included and analyses conducted as part of the 2016 Health Effects Support Documents (HESD) for the Health Advisories for each compound. In the identification of relevant studies, EPA conducted broad literature searches focused on the chemical name/synonyms with no limitations on lines of evidence (Appendix A of the documents). Therefore, any relevant study published since the 2016 HESDs should have been identified.*

EPA has also noted in both the PFOA and PFOS documents that all studies relied upon for quantitative analysis were not put through the same systematic review process. Many of the epidemiological and animal studies are qualitatively incorporated into this assessment based on the HESD. Specifically, EPA notes that only the animal studies supporting the candidate Reference Doses (RfDs) derived in the 2016 HESDs were incorporated into the systematic review methods outlined in the current SAB External Peer Review Draft MCLG documents. EPA notes that all other studies referenced from the 2016 HESD adhered to the specific criteria for inclusion in the 2016 HESDs, but study confidence between the studies included in the 2016 HESD and this assessment cannot be compared. Therefore, only the animal studies supporting the candidate RfDs derived in the 2016 HESD were considered quantitatively in this assessment.

It is important that all of the studies relied upon quantitatively be put through the same evaluation process to determine study confidence and quality. Because processes have changed since the 2016 HESDs, the requirements for a high-quality study may have changed. EPA (2021a, b) indicates that the current systematic review processes have been applied to the animal studies being used quantitatively and this would also be important for the epidemiological studies. The epidemiological studies were the focus and played an integral role in the assessment, with the animal studies

providing support. In reviewing all of the studies considering quantitatively, it is also important for EPA to confirm that the majority of the epidemiological studies relied upon for quantitative analysis in the current SAB External Peer Review Draft documents are more recent than the 2016 HESDs and therefore, should have been put through the same systematic review processes. Because there are a limited number of epidemiological studies that are pre-2016, it would not be a large effort to put these studies through the same critical review to ensure that all confidence ratings are comparable.

Noncancer Hazard Identification

1. Please comment on the health effect/outcome categories identified from the review of the available literature. Do you agree with the strong vs. suggestive evidence designations for the various health outcome categories? Do any other health systems or endpoints need to be considered for POD derivation?

Comment: *The most striking aspect of the EPA review of the health effect/outcome categories identified from the PFOA and PFOS literature is that, while the evidence is characterized as suggestive for many endpoints, there is only one health outcome where the EPA characterizes the evidence as strong: an apparent association of PFOS exposure and Total Cholesterol/LDL-C (but not directly with CVD). Even in this case, the PFOA document does not characterize the association as strong. These equivocal characterizations reflect the fact that, despite the large number of studies that have been carried out on PFOA, the evidence from animal studies is almost exclusively from studies conducted with dosing in the range where effects may be associated with activation of PPAR-mediated disruption of lipid metabolism, which is not relevant to the much lower exposures experienced by human populations. On the other hand, the evidence from epidemiological studies is highly susceptible to confounding by pharmacokinetic interactions between the health outcome being studied and the transport/excretion of PFAS (Andersen et al. 2021a,b, open access).*

Reduced birthweight provides an example of the potential for unrecognized confounding in epidemiological studies with PFAS. As cited by EPA, numerous studies of human populations have reported small decreases in birth weight in relation to increasing PFOA and PFOS blood concentrations. However, the C8 Science Panel evaluated the epidemiological evidence in 2011 and concluded that there was not a probable link between exposure to PFOA and low birth weight (C8 Science Panel 2011; Stein et al. 2009; Savitz et al. 2012; 2011b). Since then, additional epidemiological studies have reported small reductions in infant birth weight (less than 20 grams per ng/ml increase of PFOA or PFOS) (ATSDR 2021). Steenland et al. (2020) recently reviewed literature published since 2011. They attributed the association between PFOA or PFOS and decreases in birth weight as possibly due to reverse causality or confounding. Studies with insufficient exposure contrast (i.e., low exposures with little variability), such as those in the general population (in the absence of drinking water contamination) are particularly prone to distorted effects due to reverse causality and confounding. Stronger associations with birth weight are seen in studies when PFOA is measured later in pregnancy (Steenland et al. 2018; Apelberg et al. 2007; Chu et al. 2020). When PFOA is measured earlier in pregnancy, the associations with birth weight are largely null (Darrow et al. 2013; Manzano-Salgado et al. 2017; Steenland et al. 2018). In addition, most epidemiological studies that specifically evaluated the risk of low birth weight (that is, birth weight <2500 grams) have reported null associations with increased concentrations of PFOA or PFOS (Savitz et al. 2012a; 2012b; Darrow et al. 2013; Stein et al. 2009; Chen et al. 2012; Manzano-Salgado et al. 2017).

These findings are consistent with the hypothesis that the apparent association between PFOA and birth weight is confounded by the magnitude of plasma volume expansion during pregnancy and glomerular filtration rate (Steenland et al. 2018; Verner et al. 2015). Evaluations of potential pharmacokinetic bias have demonstrated that associations between prenatal exposure to PFAS and lower birth weight in epidemiological studies may actually be driven by changes in glomerular filtration, which increases by about 50% during the first half of pregnancy followed by a slight decline in the second half (Andersen et al. 2021a). Studies have shown that women with less of an increase in GFR tended to have smaller babies (Gibson, 1973; Morken et al., 2014). Verner et al. (2015) used a PBPK model to run simulations of a study population and to generate pairs of predictions for PFAS level and birth weight. Results obtained from simulated PFAS levels and birth weights were compared with published epidemiological studies to evaluate how much of this association might be attributable to the influence of GFR. The analysis used a detailed PBPK model of PFOA and PFOS during pregnancy (Loccisano et al., 2013) that was modified to describe the association of GFR with birth weight. The model was then used to simulate study populations exposed to PFOA or PFOS and to predict the resulting distributions of concentrations in maternal and cord plasma. Results from Monte Carlo PBPK model simulations (of longitudinal data) indicated that even controlling just for the effect of GFR changes accounted for the majority of the association of PFOA and PFOS with reduced birth weight.

2. Elevation of liver serum biomarkers in humans is frequently used an indication of liver injury, although it has not been shown to be as specific as functional tests, such as histology findings and liver disease (Boone, 2005, HERO ID: 782862). However, greater than 2-fold increases in alanine aminotransferase (ALT) activity, the most sensitive test of hepatocellular injury in humans, above the upper limit of normal are considered indicative of hepatocellular injury. EPA concluded that the available data in adults show a consistent positive association between PFOA and/or PFOS exposure and increased serum ALT levels in the epidemiological literature. However, this response was not selected for dose response modeling because 1) the magnitude of the effect was not large compared to control levels; and 2) concerns about the clinical relevance of the findings and non-specificity of the biomarkers relationship to adverse liver injury and disease.

Comment: *EPA concerns regarding the clinical relevance of the small increases in ALT reported in the epidemiological literature and the non-specificity of ALT as a biomarker of liver injury/disease are justified. They would be not be appropriate as a basis for setting a quantitative exposure guideline.*

3. Does the SAB panel agree with EPA's rationale for not considering the ALT endpoint reported in the epidemiological studies for the derivation of a POD for the liver health effects? Please provide your justification and if you suggest that EPA consider this endpoint for POD derivation, please provide your recommendations for a modeling approach.

Comment: *The decision not to select ALT for dose-response modeling is appropriate due to the questionable clinical relevance of the small increases in ALT reported in the epidemiological literature and the non-specificity of ALT as a biomarker of liver injury/disease particularly. The relevance of these observations of small changes in ALT is of particularly questionable in the case of PFAS, due to the likelihood that the associations may be secondary to a reverse relationship between altered liver function and the control of PFAS transport/excretion, where individuals with impaired liver function may have reduced transport/clearance of PFAS, resulting in relatively higher blood*

concentrations compared to healthy individuals (similar to the case of impaired kidney function, described in the comment on cancer classification below).

A. Are you aware of additional studies that support the ALT levels as markers of adverse liver effects? Please provide citations.

Comment: *No.*

B. Are there other adverse liver endpoints identified in the epidemiological literature that need to be considered?

Comment: *No.*

Cancer

1. Cancer classification for PFOA/PFOS

A. PFOA: Based on new cancer studies identified since the 2016 PFOA Health Advisory (HA), EPA concludes that the available cancer data for PFOA indicate a 'likely carcinogen' categorization which is a change from 'suggestive' in the 2016 HA. Does the panel agree with the 'likely' designation based on the new evidence? If yes, is the rationale clearly described? If no, please provide an explanation for arriving at a different conclusion.

Comment: *The EPA's proposed change in the categorization of PFOA from "suggestive evidence" to "likely carcinogen" is not justified. The EPA's determination appears to be based on an epidemiology study reporting an association between PFOA concentrations and incidence of renal cell carcinoma (Shearer et al. 2021). PFOA (median 5.5 ng/mL sampled during 1993-2001) was measured in blood serum at least 2 to 18 years before diagnosis of kidney cancer; given the half-life of <2 years (see comments under Toxicokinetics section), a single PFOA measurement is unlikely to accurately portray the exposure relevant to the development of disease. In epidemiological studies of higher exposures, there is inconsistent evidence of increased kidney cancer risk. Epidemiological studies of residents exposed to PFOA and other PFOS in contaminated drinking water have reported modest increases in kidney cancer (Li et al. 2022; Vieira et al. 2013). Studies of occupational cohorts have been inconsistent, with one cohort showing decreased risk of kidney cancer (Raleigh et al. 2014) and another cohort showing increased risk of kidney cancer (Steenland and Woskie, 2012); however, the number of kidney cancer cases or deaths in the occupational cohorts have been relatively few, and the investigators have cited low statistical power to draw conclusions regarding the reported associations. Nevertheless, if there were a strong causal association between PFOA or PFOS exposure and kidney cancer, it would be expected that much higher estimates of relative risk (a magnitude of 3-fold or more) would be seen in occupational cohorts who were exposed to PFOA at much higher concentrations than the general population and who were followed for more than 30 years on average (Raleigh et al. 2014; Steenland and Woskie 2012). Separately, kidney cancer is frequently associated with impaired kidney function. Lower renal function (calculated as estimated glomerular filtration rate (eGFR)) is likely to result in decreased PFOA excretion and a consequent increased concentration in serum. Cross-sectional analyses of adults exposed at background levels (Shankar et al. 2011) and of children exposed at high levels (Watkins et al. 2013) found a positive*

association between lower kidney function (i.e., lower eGFR) and higher measured serum PFOA. Dhingra et al. (2016), performed an analysis of cross-sectional studies reporting associations between PFOA and renal function, and concluded that reverse causation led to the observed associations. Shearer et al. (2021) reported that a higher percentage of cases (9%) than controls (5.6%) had diminished kidney function; however, the overall difference in kidney function between cases and controls was not statistically significant when kidney function was stratified by normal (eGFR ≥90 mL/min/1.73 m^2), mild loss (eGFR 60-89 mL/min/1.73 m^2) or diminished kidney function (eGFR <60 mL/min/1.73 m^2). In sensitivity analyses, Shearer et al. (2021) stratified by kidney function and separately restricted analyses to study subjects with high kidney function. In both analyses, the odds ratios for kidney cancer were statistically significantly increased for PFOA exposure; however, it is not clear that eGFR based on a single sample collected 2 to 18 years before the diagnosis of kidney cancer falls within a relevant time window for kidney cancer induction and latency associated with impaired kidney function. Steenland and Vieira (2021) reviewed these studies (including the Shearer et al. 2021) and concluded that the evidence from epidemiologic studies of PFAS in relation to cancer "remains limited."

B. PFOS: Based on a small number of new cancer studies identified since the 2016 PFOS HA, EPA concludes that the available cancer data for PFOS indicate a 'suggestive' categorization which is unchanged from the categorization identified in the 2016 HA. Does the panel agree that the new studies do not change the designation? If yes, is the rationale clearly described? If no, please provide an explanation for arriving at a different conclusion.

Comment: *The decision to continue the 'suggestive' categorization is appropriate. Although Shearer et al. (2021) also reported statistically significantly increased odds ratios for kidney cancer when PFOS was measured as a continuous variable, the odds ratios did not increase with increasing exposure when PFOS exposure was categorized. After adjusting for PFOA and PFHxS, there was a non-significantly decreased odds ratio for kidney cancer (OR 0.92, 95% CI 0.60–1.42).*

2. Cancer Slope Quantification: EPA used the Shearer et al., 2021 epidemiological study to quantify a cancer slope factor using peak exposure for PFOA. Has EPA adequately justified the use of this study and peak exposure for the quantification of a cancer slope factor for PFOA? If no, please describe alternate approaches that SAB recommends.

Comment: *The EPA relies extensively (and exclusively, for epidemiological studies generated since the 2016 HESD) on the Shearer et al. (2021) study. However, there are earlier studies of occupational groups exposed to much higher exposures that were discounted largely because of small numbers of cases and questions about whether the studies had adequate power to detect an excess cancer risk if one existed (Steenland and Woskie et al. 2012; Raleigh et al. 2014). See response to Cancer Classification.*

(i) Use of Shearer et al. (2021) study: For purposes of deriving the cancer slope factor, the EPA estimated the dose-response between PFOA and kidney cancer using a weighted linear regression of the quartile-specific odds ratios where the weights were inverse variance of each OR. Although Shearer et al. (2021) reported a statistically significant increased risk of renal cell carcinoma when exposure was modeled as a continuous variable. In a separate categorical analysis, an exposure-response relationship was not seen, that is, the odds ratios did not increase with increasing exposure. In the analysis that adjusted (OR 1.71, 95% CI 1.23–2.37) after

adjusting for body mass index, smoking status, history of hypertension, estimated glomerular filtration rate, previous freeze-thaw cycle, and calendar year of blood draw, the OR was statistically significant for the highest quartile of PFOA exposure only.), a separate categorical analysis did not show an exposure-response relationship. The odds ratios did not increase with increasing exposure, although the OR was increased for PFOA in blood serum concentrations >7.2 to 27.2 ng/mL when compared to concentrations <4.0 ng/mL. After further adjustment for exposure to other PFAS, the OR for the 4th quartile was attenuated, and was not statistically significant. (see Table below). The p-for-trend was 0.13 (not significant). The Shearer et. Al. (2021) study should not be used to derive a POD for calculating a cancer slope factor.

Shearer et al. (2021)

Cases / Controls	ng PFOA/mL blood	OR	95% CI
47 / 81	<4.0	1.00	Reference
83 / 79	≥4.0 – 5.5	1.41	0.69–2.90
69 / 83	>5.5 – 7.3	1.12	0.52–2.42
125 / 81	>7.3 – 27.2	2.19	0.86–5.61
	Continuous	1.68	1.07 – 2.63

Study participants were 55-74 years at time of blood draw. Serum samples were collected at a single point in time (before diagnosis of kidney cancer). Kidney cancers were diagnosed on average 8.8 years after the blood draw (range, 2-18 years). In comparison, Steenland and Woskie (2012) studied mortality among 5,791 workers exposed to PFOA during 1952 to 2004. These workers had mean duration of employment of 19 years, a mean duration of follow up of 30 years, and an estimated mean annual serum concentration of 350 ng/mL. When kidney cancer mortality was compared to other workers in the same region, there was no exposure-response relationship when PFOA was categorized according to quartiles of cumulative PFOA exposure. Although an excess of kidney cancer deaths was seen for workers exposed to ≥1,819 ng/ml-years (the 4th quartile) when exposure was lagged by 20 years to account for latency, the SMR was decreased for the second quartile and there were no kidney cancer deaths that occurred in the third quartile of exposures (Steenland and Woskie, 2012). Although some have argued that the results from the worker studies are limited because workers are healthier than adults in the general population, the healthy worker effect is not a significant source of bias when evaluating cancers with long latencies (Checkoway et al, 2014).

Under a hypothetical assumption of 30 years of exposure to 15 ng/mL (which represents a mid-range estimate from the highest quartile of exposure (>7.3 to 27.2 ng/mL) in Shearer et al. (2021), the cumulative exposure would be 450 ng/mL-years for cases and controls in the highest quartile. These cases from the Shearer et al. (2021) population would fall within the lowest occupational exposure category from the Steenland and Woskie (2012) workers (1st quartile, 0 to <515 ng/mL-years, when exposure was lagged 20 years to allow for an appropriate induction and latency for kidney cancer). Among workers exposed to PFOA, three kidney cancer deaths were reported (compared to 2.2 expected, SMR 1.34, 95% CI 0.28 – 3.91) (Steenland and Woskie et al. 2012). Raleigh et al. (2014) relied on a job-exposure matrix to estimate individual concentrations of APFO in micrograms per cubic meter of air, rather than drawing blood for analysis. Although these estimates of inhalation exposure reported by Raleigh et al. (2014) cannot be directly converted into serum PFOA concentrations, other studies of workers that overlapped with the Raliegh et al. (2014) cohort (Olsen et al. 2003; Olsen and Zobel, 2007) reported very high serum concentrations of PFOA in the blood of workers (>1,000 ng/mL) and at concentrations higher than those reported by Steenland and Woskie (2012). In contrast to

Steenland and Woskie (2012), Raleigh et al. (2014) did not find an increased risk of kidney cancer in the highest category of exposure (4th quartile, hazard ratio (HR) 0.73, 95% CI 0.21-0.48) when compared to the non-exposed population based on four incident kidney cancers. There was also no increased risk of kidney cancer when the 3rd and 4th quartiles were combined (8 kidney cancers, HR 0.85, 95% CI 0.36–2.06).

Shearer et al. (2021) chose quartile cut-points based on the serum PFOA exposure in the controls. There was, however, a substantial difference in the number of kidney cancer cases (n=47) and controls (n=81) for the lowest exposures (1st quartile, <4.0 ng/mL). This concentration is similar to the geometric mean concentration for PFOA (4.8 ng/mL for participants age 60 years and older) in a nationally representative sample (NHANES) collected during 1999-2000. It seems unlikely that there would be a discrepancy in kidney cancer incidence in a population exposed at background levels and there is no logical explanation for a deficit in kidney cancers at these low concentrations. The clear discrepancy between the number of cases and controls in the referent category may potentially create a spurious association between PFOA and kidney cancer when higher quartiles of exposure are compared to the referent group (1st quartile of exposure).

The Shearer et al. (2021) study does not consider cancer induction and latency, which is a study limitation, especially considering population-level exposures. Presumably, high exposures in occupational groups would result in a shorter latency than population level exposures; however, inconsistent associations between PFOA and kidney cancer have been reported in workers exposed to much higher concentrations and followed for 30 years or more (Raleigh et al. 2014; Steenland and Woskie, 2012). Solid-tumor cancers are unlikely to have cancer induction and latency periods that are shorter than 20 years at higher exposures. For example, Smith et al. (2018) reported latency periods for kidney cancer were at least 20 years following exposure to arsenic in drinking water and remained elevated after 40 years of follow up. Because the time to diagnosis of kidney cancer ranged from 2 to 18 years since blood in Shearer et al. (2021), an inadequate latency for cancer development raises the possibility of reverse causality or confounding, i.e, impairment of kidney function associated with renal cell cancer induction results in increased concentrations of PFOA. Dhingra et al. (2017) reported that decreased renal function (lower glomerular filtration rate) led to higher PFOA concentrations. Decreased renal function is expected as kidney cancer develops, although it may not be noticed before clinical manifestation of kidney cancer. Although Shearer et al. (2021) adjusted their results for estimated glomerular filtration rate (eGFR), the estimate was based on a single blood serum sample, which is inadequate for an evaluation of glomerular filtration rate. Moreover, glomerular filtration is only one of the many kidney functions that control the excretion of PFOA; multiple renal transporters, including the basolateral and apical organic acid transporters and the urate transporter, are also involved in the regulation of PFOA excretion, and hence blood levels.

(i) *<u>Use of peak exposure:</u> Peak exposure was not discussed as an exposure metric in Shearer et al. (2021), and a search of the EPA PFOA MCLG document did not identify any statement to indicate that peak exposure was used in the EPA analysis. While the California Environmental Protection Agency's Office of Environmental Health Hazard Assessment (OEHHA) used peak concentration in their derivation of a Public Health Goal (PHG) for PFOA, and it seems likely that the EPA followed a similar approach, there is no citation to the OEHHA PHG document in the EPA MCLG document. The*

uncertainty regarding the approach taken by EPA represents a serious lack of transparency in the document.

The use of peak exposure is not appropriate for a chemical like PFOA, which has a half-life on the order of a year or more. With such a long human half-life, it can take as much as a decade of continued exposure at a given intake to produce a corresponding change in target tissue concentration, so the peak internal exposure would not be well represented by the peak intake. There is also no rationale for using peak exposure as the dose metric for renal cell carcinoma from human exposures to PFOA on the basis of potential nonlinear pharmacokinetics or pharmacodynamics of PFOA over the range of human exposures. Therefore, the time-weighted average exposure should be used as the dose metric for the analysis, or cumulative exposure (that is, the product of intensity and duration) as used by Steenland and Woskie (2012).

Toxicokinetic Models

1. Human model –

   A. For endpoints observed in adults, EPA used a steady-state approach to calculate the HED, which assumes a relatively constant exposure and clearance during adulthood. Please comment on this method of HED calculation. Are there alternative approaches that EPA should consider? If so, please describe the rationale for recommending this approach(es).

   Comment: *The use of steady-state to calculate HED is an acceptable approach for cross-species extrapolation of the key model derived points of departure in this document. For longer half-lived PFAS compounds like PFOA and PFOS, it is unlikely that event-based approaches would yield different results. This may not be the case for PFAS with shorter half-lives.*

   B. Two key parameters are the half-life and volume of distribution, which were used to calculate clearance. Half-life and volume of distribution were assumed to be constant across sex and age groups because of a lack of strong quantitative data to parametrize changes across sex and age. Please comment on the strengths and weakness of the use of this assumption and the choice of these parameters by the EPA. Please describe the rationale for alternative recommended approaches. For endpoints observed in human neonates or children, EPA used a one-compartment TK model to simulate dosimetry during pregnancy and a two-compartment TK model (one-compartment models for the mother and the child) to simulate dosimetry during lactation, to calculate the HED for each POD. Please comment on the strengths and weaknesses of this choice of model structure for the task of predicting dosimetry in the human fetus and child compared to dosimetry in mice and rats in the similar lifestages. Please provide the rationale for any alternative recommended approaches.

   Comment: *The use of Thompson et al. (2010) for the Volume of Distribution (VD) for PFOA and PFOS is appropriate for human and the animal. However, the half-lives for humans used by EPA for the clearance calculation was 3.8 years for PFOA and 5.4 years for PFOS, derived from studies of retired workers (Olsen et al. 2007). Given the extensive discussion in the MCLG document of additional studies reporting the half-life of PFOA in environmentally exposed populations, it is unclear why the half-life from an occupational study that included*

only a small number of retired workers was chosen over other reported values for larger populations exposed at environmental levels, including the more accepted half-life value of 2.3 years reported by Bartell et al. (2010) study. The Zhang et al. (2013) study, which included the collection of urinary data to support an estimate of renal clearance, and which was reviewed in the EPA's half-life discussion, also appears to support the results of the Bartell that the 3.8 year half-life is incorrect. These studies provide a strong argument against using the half-life reported in Olsen et al. (2007), which is thought to include subjects that intermittently had occupational level exposures after retiring.

A recent review of studies with human PFOA half-life information by an international collaboration sponsored by the Alliance for Risk Assessment (Campbell et al., 2022) concluded that the results from the most appropriate studies support a PFOA half- life in the range of 0.5 to 1.7 years, indicating that the half-life of 3.8 years (Olsen et al. 2007) is much too high. Verner et al. (2016) did not justify the use of 3.8 years nor did they evaluate the impact of the shorter estimated half-lives in their analysis and noted that the "much greater ratio of estimated intakes of PFOA may be partially due to the half-life we used (3.8 years); others have suggested a lower value (e.g., 2.3 years)." The Agency discusses this in their half-life review but does not provide reasoning as to why they nevertheless used the 3.8-year half-life from the model in Verner et al. (2016). A broader evaluation of the half-lives reported for PFOA would indicate that, when accounting for continued exposure, the half-life is likely to be below 2 years. The Olsen et al. (2007) study also serves as the basis for the PFOS half-life used in the Verner et al. (2013) model by the EPA; it is likely that the correct half-life for PFOS is also much shorter than the value used by EPA.

C. The key chemical-specific parameters that describe the transfer of the chemical from the mother to the child during gestation and lactation are the maternal to fetal serum ratio and the ratio of maternal serum to milk PFOA/S concentration. These ratios were assumed to be constant during gestation and lactation, respectively. Another important parameter is the rate of milk ingestion, which is chemical-independent and varies throughout lactation. Please comment on the strengths and weaknesses of the choice of parameters for fetal to maternal partitioning and partitioning into breastmilk, as well as the choice for lactation rate. Please also comment on the choice to assume that fetal to maternal partitioning and partitioning to breastmilk did not vary in time. Please describe whether there are other methods you would recommend to account for these changes over time and across development.

Comment: *The use of the serum to milk and serum to fetus ratios is a default approach that, while increasing uncertainty in the fetal/lactational modeling, is not unreasonable. The application of the model to the available time-course data in the rat during lactation indicates that the approach was valid for PFOA and PFOS.*

2. Animal Model –

A. After a review of the available toxicokinetic models for PFOA/S predictions in laboratory animals, EPA selected the Wambaugh et al. (2013) model because it was parametrized using all species of interest, demonstrated good agreement with training and test datasets, and used a single, biologically motivated, model structure across all species. Does the panel agree with selecting this model? If not, please describe the rationale for alternative recommended approaches for the calculation of the internal dose metrics in adult animals.

Comment: *The choice of a simple PK model that accounts for saturable resorption is reasonable, and the approach taken (e.g., choice of dose metrics) in deriving the animal dose metrics, as well as the PODs in humans, are reasonable. Based on the model fits presented in the PFOA and PFOS documents, the models provide acceptable fits to the data overall. However, neither the animal nor the human PK model files were available to allow verification of the EPA's implementation of the published models that they actually used to derive the dose metrics. The EPA routinely verifies the code of submitted models, and the same possibility should be provided to the public for agency model code. All model files (including R scripts) should be available for review by the SAB, as well as the public, to provide full disclosure.*

B. The animal model parameters were obtained through a Bayesian inference parameterization which produced wide credible intervals for some parameter values, but relatively tight credible intervals for the predicted serum concentration. Does the panel agree with using the median values of the estimated animal parameter distributions for prediction of serum concentration and internal dose metrics?

Comment: *Median values from Bayesian inference using a PK model are reasonable to the extent that the PK model adequately captures the kinetic data used in the calibration and the implementation of the models by EPA are error-free. (see previous).*

C. Based on visual inspection of model predictions to the calibration datasets, EPA utilized sex-independent parameters for PFOS. The male-specific parameters were used for all rat-specific PFOS predictions including predictions in pregnant and nursing dams and the female-specific parameters were used for all mouse-specific PFOS predictions because the parameter values obtained from fitting the female-specific rat data and male-specific mouse data were not consistent with the overall TK parameters for PFOS and produced poor fits to the training and test datasets. Does the panel agree with this approach and justification for this assumption for PFOS? If not, please describe other approaches that could be considered?

Comment: *While it is reasonable to suggest that the kinetics of PFOS in male and female rat are similar, the impact of that assumption on the simulations of the Chang et al (2012) study presented in Wambaugh et al. (2013), and of the Kim study shown in supplemental E (Figure E-7 left panel) should be presented. Wambaugh et al. (2013) did not show the model fit to the female rat and the agency has not included simulations using the Wambaugh et al. (2013) model in Appendix E. Wambaugh et al. (2013) did not discuss their use of gender-specific parameters for PFOS in the rat, which had not been required in the previous modeling efforts. It is not possible to fully evaluate whether there is an error in the approach taken with the PFOS animal modeling, because there is a less than full documentation of the EPA model.*

D. EPA assumed a one compartment model for the developing infant based on the lack of infant-specific toxicokinetic data from rats and mice. This model utilizes averages of half-life and volume of distribution from the literature coupled with physiologically relevant lactational parameters for pup nursing. Does the panel agree with the decision to use this model structure for infant animals? If not, please provide data on infant-specific changes during the animal lactational-period that could be used to account for toxicokinetic differences between the adult and infant rats and mice.

Comment: *The model structure for the nursing pup is reasonable. The potential issues noted for the adult animal; however, could lead to issues with the prediction of fetal dose metrics. Given the adult model cannot be fully evaluated due to limited documentation, the pregnancy and lactation model can also not be fully evaluated to determine whether or not the results are correct.*

E. Several parameters dictate the transfer of chemical from the mother to her pup. Does the panel agree with the selection of these parameters for the animal model? If not, please provide your justification and alternative parameters.

Comment: *The source for the milk ingestion for animal pups was not fully documented and has not been peer reviewed – cited as in prep (Kapraun et al. 2021). The PFOA and PFOS MCLG documents should provide the information on milk ingestion by mouse and rat pups, describe whether the values are consistent or different from those used in previous lactational modeling in mouse and rat, and justify the choice of intakes. There was no documentation provided on the lactational transfer in humans and very little information is provided regarding the Verner model. While Verner et al. (2016) includes a model file in his manuscript, this does not preclude the agency from providing their model files for the SAB review, as well as for the public, since the parameters actually used in the model scripts determine the output, not the parameters in the publication.*

F. For neonatal animals, EPA assumed no sex differences in clearance in neonatal animals based on the lack of identification of sex-dependent differences in PFOA/S toxicokinetics from the available data. Does the panel agree with this assumption? If not, please provide your justification and available data on sex differences in neonatal rats.

Comment: *Sex differences in PFOA clearance have only been observed for adult rat where the female rat exhibited a much shorter half-life than the male rat. The evidence supports an assumption that PFOA and PFOS clearance in neonatal animals is similar to adults, and that it is similar to the adult female rat for PFOA.*

Epidemiological Study RfD Derivation

1. EPA evaluated potential confounding as part of their study quality evaluation of the epidemiological studies and selected only 'medium' and 'high' quality studies for POD derivation. Have the epidemiological studies that were selected for dose-response modeling sufficiently addressed confounding? If not, are there key additional analyses that could be performed to further address the potential confounding of PFAS exposures in these studies?

Comment: *The EPA appropriately evaluated confounding as a risk of bias domain when evaluating the quality of an individual study. Based on a visual review of the heat maps, it appears that a study can be deemed deficient with respect to confounding and still be judged to be a "medium" quality study. It is not clear if each of the risk of bias domains were considered of equal weight when judging the study confidence level (high, medium, low, or uninformative) or if there were certain domains that were given greater weight when assessing study confidence. A written protocol would provide guidance for the evaluation of study quality.*

In the overall synthesis of evidence from epidemiological studies, the EPA appeared to focus primarily on co-exposures to other PFAS. Potential confounding from exposures to other environmental contaminants is only discussed by citing statements in the study publications that exposures to a particular compound (e.g., PCBs) were not highly correlated with PFAS exposure. However, an evaluation of the correlation between the two exposures is not a substitute for including the co-exposures in the analysis as an additional covariate. Previous studies for other compounds by some of the same investigators have failed to include important covariates out of concern that there might no longer be a significant association for the main effect. Given the importance of these studies for the risk assessment, the EPA should not finalize this document without obtaining the data from the critical studies and performing their own analysis, so that it would be available for public scrutiny.

Another potential mechanism of confounding is referred to as pharmacokinetic (PK) bias. PK bias due to confounding arises when a confounding factor affects both the biomarker (e.g., PFAS blood concentration) and the health outcome (e.g., decreased birthweight). PK bias can also result from reverse causation, that is, when the health outcome alters biomarker levels. Importantly, PK bias analysis, whether for reverse causality or effects of confounding factors, can readily be conducted with pharmacokinetic models to examine the influence of confounding factors or health outcomes on pharmacokinetic processes and the resulting epidemiological associations (Andersen et al. 2021a). EPA has the capability to review published studies of PK bias and to perform PK bias analysis themselves, as evidenced by their previous applications of PBPK and BBDR modeling.

2. Studies of developmental immune health outcomes (Grandjean et al., 2012 [HERO ID: 1248827]; Grandjean et al. 2017 [HERO ID: 3858518]; Grandjean et al., 2017 [HERO ID: 4239492]; and Budtz-Jorgensen and Grandjean, 2018 [HERO ID: 5083631]) after PFOA/S exposure identified associations with very low doses of either PFOA or PFOS with developmental immune effects. The RfD for this outcome was selected as the critical effect because it was the lowest among the candidate RfDs for PFOA or PFOS and can result in severe illness. Does the panel agree with the selection of the critical study and critical effect for the derivation of chronic RfDs for PFOA and PFOS?

   A. If so, please explain your justification.

   B. If not, please provide your rationale and detail an alternative critical study and/or critical effect you would select to support the derivation of chronic RfDs.

   C. Are any additional analyses or rationales needed to increase the confidence in the chronic RfDs for PFOA and PFOS?

Comment: *The EPA stated that human epidemiological studies consistently reported decreases in antibody response following vaccination and recommended antibody response to vaccination in children as an outcome for POD derivation. For the POD derivation, the EPA relied an analysis of response to tetanus and diphtheria vaccination in two birth cohorts of children in the Faroe Islands (Budtz-Jorgensen and Grandjean, 2018). Should the EPA decide to use the Faroe Island studies as the basis of the RfD, the data should be obtained from the study investigators and independently evaluated. The analysis and interpretation of the results from the series of studies deserves further review due to inconsistencies within and between the studies regarding participants and methods. Additional concerns are provided below.*

<u>Level of clinical protection:</u> Grandjean et al. (2012) reported 2-fold and 4-fold increased ORs for falling below a clinically significant protective level for tetanus and diphtheria antibodies at age 5 years and age 7 years, respectively. The clinically protective level used by the study investigators was 0.1 IU/mL. Grandjean et al. (2012) reported: "[s]erum concentrations of antibodies against the tetanus toxoid were measured in coded samples by the Statens Serum Institut using enzyme-linked immunosorbent assay (Hendriksen et al. 1988)." Hendriksen et al. (1988) describes the toxin binding inhibition test (ToBI), an assay which is a modified ELISA, for which clinical protection is achieved at 0.01 IU/mL (WHO, 2018; WHO, 2017).

The WHO (2017) reports:

> *"There is no definitive immunological correlate of protection for tetanus. The minimum amount of circulating antibody that, in most cases, ensures immunity to tetanus is assay-specific. Using in vivo neutralization tests or modified enzyme-linked immunosorbent assays (ELISA), concentrations exceeding 0.01 IU/ml are usually considered protective, whereas antibody concentrations of at least 0.1–0.2 IU/ml are defined as protective when using standard ELISA techniques. [p. 61]"*

The WHO (2018) states:

> *"A toxin binding inhibition (ToBI) assay has been reported and demonstrated to show good correlation with the neutralization assay (correlation coefficient = 0.95) (Hendriksen et al., 1988). The assay determines the level of inhibition of binding of TT to a polyclonal antitoxin by tetanus antibodies in the test sera. The ToBI assay has been subsequently demonstrated to be able to measure tetanus antibody levels below 0.01 IU/ml, making the test attractive for assessing tetanus immunity (van Gageldonk et al., 2008). [p.7]"*

For vaccination against tetanus and diphtheria, the children in the Faroe Islands cohorts followed a vaccination schedule of 3 shots in the primary series (at ages 3 months, 5 months, and 1 year) and one booster at age 5. The current recommendation by the WHO for primary vaccination and booster doses in children recommends a primary series of 3 doses of TTCV (similar to the Faroe Islands schedule for primary series) and a boosting regime of 3 doses of TTCV for a total of 6 doses in order to achieve long-term immunity (WHO, 2017):

> *"The 3 TTCV booster doses should be given at: 12–23 months of age; 4–7 years of age; and 9–15 years of age. Ideally, there should be at least 4 years between booster doses."*

The duration of protection and requirements for booster is further illustrated in the WHO Immunological Basis for Immunization Series, Module 3: Tetanus (WHO, 2018):

> *"To illustrate the kinetics of immunity among children ≥1 year, adolescents and adults following primary and booster vaccination with TTCV, Figure 2 provides a schematic diagram of the typical response. A single dose of TT in the absence of priming induces little, if any, protection. Two to four weeks after the second dose, the mean level of tetanus antitoxin typically exceeds the minimum putatively protective level of 0.01 IU/mL. One year after the second dose, the mean antibody levels are expected to decline and may fall to the protective threshold level. After each subsequent dose of vaccine, immunity is boosted, then persists above the protective threshold for a time, and then wanes over time. Putatively protective levels of immunity are induced by a primary series of three TTCV doses and immunity typically persists for at least 5 years. After the third dose, each additional booster dose given after at least a one-year interval increases tetanus antitoxin levels and further prolongs the duration of immunity. Immunity may persist for approximately 10 years after the fourth dose of TTCV and for at least 20 years after the fifth dose. [WHO, 2018, pp. 14-15]."*

Figure 2: Schematic diagram of the antibody response to tetanus toxoid (TT) among children ≥1 year, adolescents and adults

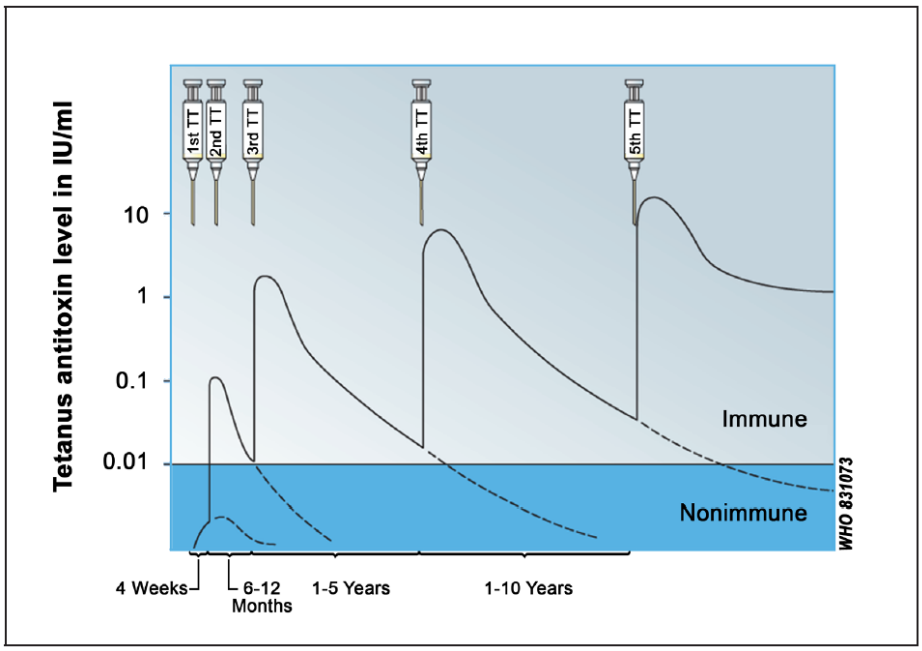


Figure reproduced from: Plotkin S, Orenstein W, Offit P, Edwards KM. Plotkin's vaccines, seventh edition. New York (NY): Elsevier; 2017: Chapter 58, Tetanus toxoid, p. 1071, reproduced by permission.

"Five properly spaced doses [given to children ≥1 year, adolescents and adults provides] protection lasting at least 20 years and probably substantially longer for most recipients. (Threshold of protection of 0.01 IU/mL applies to values measured by in vivo neutralization, modified ELISA [or bead-based immunofluorescence]; for standard ELISA, protection is [usually defined between 0.10–20 IU/mL, based on the assay])." (Modified source: Borrow R, Balmer P, Roper MH. The immunological basis for immunization. Module 3: Tetanus (update 2006). Geneva: World Health Organization; 2007.)

Although the WHO (2012) Guidelines for Immunotoxicity Risk Assessment recommend measures of vaccine response as a measure of immune effects, the guidelines refer to EHC 180 (UNEP, 1996):

> *"A description of biomarkers in epidemiological studies is provided in EHC 180: Principles and methods for assessing direct immunotoxicity with exposure to chemicals (IPCS, 1996). The risk assessor should refer to the assay descriptions in EHC 180 for immunotoxicity end-points contained in the data set for the chemical in question to provide specific context, cautions and information that may assist in the interpretation of immunosuppression data for risk assessment. In addition, it is recommended that the risk assessor consult an expert in immunotoxicology or clinical immunology to help interpret the biological plausibility of the study results. [p.51]"*

EHC 180 Principles and methods for assessing direct immunotoxicity with exposure to chemicals (UNEP, 1996) described assays recommended by WHO (UNEP, 1996) and the National Research Council (NRC) of the National Academy of Sciences (NRC, 1992) for preliminary assessment of individuals exposed to immunotoxicants, including secondary antibody responses to proteins (e.g., diphtheria, tetanus, poliomyelitis) and polysaccharides (e.g., pneumococcal, meningococcal). Specifically, the WHO (1996) describes the tests for antibody response to immunization:

"In order to test for T cell-dependent antibody responses, commercially available diphtheria-tetanus vaccine can be given in recommended doses. Blood is taken two weeks after each injection and tetanus and diphtheria antibodies are determined. In patients who have been immunized with diphtheria-tetanus or diphtheria-pertussis-tetanus vaccine, one booster injection is given before determination of antibodies. In testing for T cell-independent antibody responses, commercially available pneumococcal vaccine can be given in recommended doses. Three doses of killed poliomyelitis vaccine (10 ml intramuscularly, at intervals of two weeks) can also be used as the immunogen. Blood is taken two weeks after the last injection, and antibody is usually determined by virus neutralization. [WHO, 1996, pp. 238-239]"

Therefore, it seems that the most relevant findings from the studies in the Faroe Islands are the associations between 5 year post-booster responses compared to PFAS serum measured at age 5 years pre-booster, four weeks earlier. None of the studies of the Faroe Islands reported whether any person had a post-booster concentration that fell below 0.1 IU/mL four weeks after receiving a booster.

Although there is no information on the number of individuals who received a booster but failed to mount an appropriate response within four weeks, there is evidence that the booster worked overall. For tetanus, the post-booster median concentration (35 IU/mL, interquartile range (IQR) 16-96) in 456 children was 159-fold higher than the pre-booster median concentration based on 532 children (0.22 IU/mL, IQR 0.10 – 0.51) (Grandjean et al. 2012, Table 1). There is approximately the same variability in post-booster and pre-booster concentrations based on the IQR (6-fold difference from 75th to 25th percentiles post booster and 5-fold difference from 75th to 25th percentiles in antibody concentrations pre-booster). For diphtheria, the post-booster median concentration (13 IU/mL, IQR 6.4–26) was 108-fold higher than the pre-booster median concentration (0.12 IU/mL, IQR 0.05 – 0.40). There was a 4-fold difference between the 75th to 25th percentiles after boosting, and an 8-fold difference between the 75th and 25th percentiles before boosting.

<u>Inconsistencies regarding study subjects included in various analyses:</u> Although the Faroe Islands are four separate studies, they each address one or two birth cohorts (1997-2000 and 2007-2009) from a single hospital in the Faroe Islands. There are inconsistencies reported in relation to the number of participants at different ages in these cohorts. For the 1997-2000 birth cohort, Grandjean et al. (2012) and subsequent studies reported "[a] total of 587 children (89% of the cohort) participated in 1 or more of the examinations, which took place at age 5 years pre-booster, approximately 4 weeks after the booster, and at age 7 years." Separately, Grandjean et al. (2012) reported "[a] lower antibody response was observed in 2 groups of 173 and 168 children, who had been inoculated with combination booster vaccines containing pertussis, polio, or both, as compared with the 151 who received diphtheria and tetanus toxoids only." The total of 492 children receiving a booster dose does not match with the 532 children attending the age-5 pre-booster examination or the 456 children attending the age-5 post-booster exam (Grandjean et al. 2012, table 1) or the 537 children included in the 5-year pre-booster analysis or the 440 children included in the 5-year post-booster analysis (Grandjean et al. 2012, table 2).

It is also not clear that the children examined at the 5-year post-booster examination were restricted to the same children that participated in the 5-year pre-booster exam. In fact, it seems more likely that there were children who participated in the 5-year post-booster exam that were not part of the study population that had attended the 5-year pre-booster exam (and therefore, had not provided a PFAS serum sample 4 weeks earlier). Grandjean et al. (2012) reported that "[f]or the 5-year post-

booster data, we adjusted for the time since vaccination, using a restricted cubic spline (Heilmann 2006)." There would be no need to make this adjustment for time since last vaccination if all of the 5-year post-booster results were restricted to the same individuals who attended the 5-year pre-booster exam 4 weeks earlier. Potentially, some of the children at the 5 year post-booster exam were sampled for PFAS serum and antibody concentration immediately after they had their booster shot. If the 5-year post-booster serum samples described mixed populations (individuals who were last vaccinated four weeks earlier as well as individuals who were last vaccinated at one year), the analyses of 5-year post-booster data would have significant variability in levels of circulating antibodies. Regardless of PFAS serum concentrations, some proportion of individuals included in the post-booster analysis would have had relatively high levels of circulating antibodies (that is, individuals who were boosted 4 weeks earlier) while others would have had low levels of circulating antibodies (those who had last received a vaccination at age 1) due to the well-known effect of waning antibodies 1 to 3 years after the primary series (WHO, 2017; 2018). The analysis of post-booster results should be restricted to the individuals who provided serum four weeks earlier so that the boosting effect is not diluted.

<u>Confounding:</u> Many factors affect humoral immunity and vaccine response. Possible confounding due to co-exposures to PCBs and other persistent organic compounds, as well as methyl mercury (MeHg), existing in studies of the Faroese Island populations. The Faroese diet includes a high proportion of whale meat, which contains high levels of PFAS (Weihe et al. 2008), but it also contains high levels of PCBs, polybrominated flame retardants and MeHg. (Andvik et al. 2021). The PFAS exposure from eating whale meat is significant: "On a relative scale, a high intake of two pilot whale dinners per month is associated with increases in the 14-year serum concentrations of PFOS, PFNA, and PFDeA by almost 25%, 50%, and 100%, when compared to concentrations in subjects eating little or no whale at all (Table 3). Fish dinners had a much weaker effect, although each weekly fish dinner augmented the PFHxS concentration by about 10%." (Weihe et al. 2008)

Budtz-Jorgensen and Grandjean (2018) cited Grandjean et al. (2012) when they reported that confounding by methyl mercury and polychlorinated biphenyls was unlikely because of their weak correlations with serum concentrations of PFAS. The evaluation of methyl mercury as a potential confounder should be independently confirmed by EPA. Although Grandjean et al. (2012) reported that PCBs were weakly correlated with PFAS, methyl mercury was not discussed in these analyses and was presumably not part of the evaluation. The IPCS (1996) reported that MeHg decreased humoral immunity in mice in a study conducted by Blackley et al. (1980).

<u>Conclusion:</u> It seems more likely than not that the results of this study are merely consistent with heterogeneity in vaccine response. Furthermore, reverse causality and/or confounding are likely to be an issue in studies with little variation in exposure and low exposure contrasts. Exposure varied little in the Faroe Island studies; the exposure contrasts were low based on measurements of PFAS in serum. Although the study investigators did not provide minimum or maximum concentrations, they reported interquartile ranges. In children measured for PFAS in serum at age 5, the IQRs were: PFOA, 3.33 to 4.96 ng/mL (median 4.06); PFOS, 13.5 to 21.1 ng/mL (median 16.7); PFHxS 0.45 to 0.88 ng/mL (median 0.63) ; PFNA 0.76 to 1.24 ng/mL (median 1.00); and PFDA, 0.21 to 0.38 ng/mL (median 0.28). Grandjean et al. (2012) and subsequent evaluations of the Faroe Islands cohort used log base 2 transformations of these exposure data (with low exposure contrasts) and found that the strongest effect was at the lowest concentrations, which seems biologically implausible.

3. The health outcomes identified in the critical studies were decreased antibody response, specifically in serum anti-tetanus and anti-diphtheria, in children after vaccination (Grandjean et al., 2012 [HERO ID: 1248827]; Grandjean et al. 2017 [HERO ID: 3858518]; Grandjean et al., 2017 [HERO ID: 4239492]; and Budtz-Jorgensen and Grandjean, 2018 [HERO ID: 5083631]). This health outcome represents an increased susceptibility to a disease that can cause very severe symptoms, including lethality. Furthermore, children who are immunocompromised may mount a lower antibody response and in turn, be more susceptible to contracting the disease, if exposed than healthy children. Because this health outcome has the potential for severe illness and was assessed in children (i.e., EPA guidelines [US EPA, 1991] support a 5% BMR for developmental effects), a benchmark response (BMR) of 5% was selected for benchmark dose modeling. While some clinical findings are available, the clinical relevance of a 5% decrease in antibody response is not clear. Given the need to protect sensitive subpopulations (e.g., children, individuals with pre-existing conditions) and the available clinical data (i.e., antibody response clinical level), does the SAB support the 5% BMR selection for modeling to identify the POD? If not, please recommend the BMR level and a scientific rationale for an alternative selection.

Comment: *As stated in the comments on the previous question, the level of circulating antibody that correlates with clinical protection is assay-specific. Based on the ToBI assay used in the Faroe Islands, the relevant clinical level of protection for the population in that study would be 0.01 IU/mL, not the value of 0.1 IU/mL used by the EPA. Furthermore, none of the studies of the Faroe Islands population provided information on whether the prevalence of failure to respond to secondary immunization was beyond that expected from natural variation in vaccine response. In addition, epidemiological studies of PFOA and/or PFOS and common infectious diseases and their symptoms (including otitis media, common colds, gastroenteritis, respiratory tract infections, fevers) have reported inconsistent associations (Granum et al. 2013, Impinen et al. 2018, Dalsager et al. 2016; Looker et al. 2014; Okada et al. 2012), bringing into question the validity of the EPA's assumptions regarding the clinical relevance of the antibody titer endpoint.*

4. EPA has evaluated and applied where appropriate uncertainty factors to account for intraspecies variability (UFH), interspecies differences (UFA), database limitations (UFD), duration (UFS), and LOAEL-to-NOAEL extrapolation (UFL) for PFOA and PFOS.

   A. Has uncertainty been adequately accounted for in the derivation of the RfDs? Please describe and provide suggestions, if needed.

      Comment: *Yes. Since the RfDs are derived from data in human populations, an uncertainty factor of 10 to consider human interindividual variability is more than adequate to properly account for uncertainty in the derivation of the RfDs.*

   B. Does the provided scientific rationale support the application of the selected uncertainty factors? Please explain.

      Comment: *Yes. Since the RfDs are derived from data in human populations, no uncertainty factors are required apart from one for human interindividual variability.*

Relative Source Contribution

1.EPA applies a Relative Source Contribution (RSC) when calculating the MCLG to provide a margin of safety that an individual's total exposure from a contaminant does not exceed the RfD. The RSC is the portion of an exposure for an individual in the general U.S. population estimated to equal the RfD that is attributed to drinking water; the remainder of the exposure equal to the RfD is allocated to other potential sources. Based on the physical properties, detected levels, and available exposure information, there are significant potential sources other than drinking water ingestion for PFOA and PFOS; however, information is not available to quantitatively characterize exposure from these different sources. EPA followed Agency guidance on how to derive an RSC (U.S. EPA, 2000; available online at: https://www.epa.gov/sites/default/files/2018-10/documents/methodology-wqc-protection-hh-2000.pdf) and recommends an RSC of 20 percent (0.20) for PFOA and PFOS. This RSC is the same as what was used in the 2016 HAs for PFOA and PFOS.

A. Are you aware of additional relevant exposure data that EPA should consider in developing the RSCs for PFOA and PFOS? If so, please provide citations.

B. Please provide comment on whether the recommended RSC of 20 percent (0.20) for PFOA and PFOS is adequately supported and clearly described.

***Comment**: The basis for the recommended RSCs of 50% for infants/children and 20% for adults is adequately supported.*

EPA'S DRAFT FRAMEWORK FOR ESTIMATING NONCANCER HEALTH RISKS ASSOCIATED WITH MIXTURES OF PER- AND POLYFLUOROALKYL SUBSTANCES

Overall charge: EPA is seeking SAB comment on whether the framework and illustrative examples provided in the document are scientifically supported, clearly described, and informative for assessing potential health risk(s) associated with exposure to mixtures of PFAS.

General Comment: *The EPA Draft Framework is a very impressive document that proposes a scientifically sound approach and applies widely accepted practices for mixture risk assessment. It also provides informative examples that help to clarify the proposed approach. It represents a major improvement over some approaches that have been used by regulatory agencies, such as applying the PFOA RfD to total PFAS concentrations. It is important to emphasize, however, that these approaches are best suited for site- or source-specific assessments, where the composition of the mixture at the site or source has been determined. While the approaches described in the framework are sound risk assessment tools, it should be made clear that they are not as well suited for use by states or other entities in the promulgation of drinking water standards, where the composition of the mixtures of PFAS compounds in drinking water may not be consistent across locations and sources*.

Charge questions

1. The component-based mixtures approaches presented in the framework are based on dose addition. Traditionally, an assumption of dose addition for a mixture is based on components sharing a common mode of action (MOA) for a given health effect. However, EPA's supplementary guidance (EPA, 2000) states: "The common mode-of-action (MOA) assumption can be met using a surrogate of toxicological similarity, but for specific conditions (endpoint, route, duration)." This suggests that although the common MOA metric for application of dose addition is optimal, there is flexibility in the level of biological organization at which "similarity" can be determined among mixture components. As an emerging chemical class, MOA data is limited or not available for many PFAS. For purposes of a component-based evaluation of mixtures additivity for PFAS, EPA assumes similarity at the level of toxicity endpoint/health effect rather than MOA.

    A. Please comment on the appropriateness of this approach for a component-based mixture evaluation of PFAS under an assumption of dose additivity.

    Comment*: Grouping chemicals for dose-additivity on the basis of similarity in toxicity/health effects begs the question of "how similar". For example, dose additivity is not necessarily appropriate for a chemical that causes centrilobular hypertrophy and a chemical that causes single cell hepatocellular necrosis. In the specific case of PFAS, if the effects in a tissue differ somewhat, it is critical that the assumption of dose-additivity be restricted to compounds that also have sufficient structural similarity to support the likelihood that the key elements of the Mode of Action are similar (that is, similar interactions of the compounds with the key proteins controlling the pharmacokinetics and pharmacodynamics of PFAS). As in the case of pyrethroids and PCBs, it may be necessary to define sub-categories of PFAS for which dose-additivity can be applied. Two examples of the potential considerations are short vs. long chain length, and linear fluoroalkyl acids vs.branched-chain flouroether acids.*

B. 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).

***Comment**: Common toxicity endpoint/health effect is an acceptable similarity domain for those PFAS with limited or no available MOA-type data. However, the concerns raised in the comment on the previous question need to be adequately addressed as part of the evaluation of similarity.*

2. Section 4.3 (Hazard Index; HI) of the framework document demonstrates the application of a component-based mixture approach, based on dose addition, using available oral reference doses from completed EPA human health assessments, and hypothetical exposure information. The example calculations presented are primarily focused on four PFAS with finalized EPA Human Health Assessments: perfluorooctanoic acid (PFOA), perfluorooctane sulfonic acid (PFOS), perfluorobutane sulfonic acid (PFBS), and hexafluoropropylene oxide (HFPO) dimer acid and HFPO dimer acid ammonium salt (referred to as "GenX chemicals").

A. 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.

***Comment**: The proposed approach is reasonable and is consistent with EPA practice.*

B. Please provide specific feedback on whether the proposed HI methodologies in the framework are scientifically supported for PFAS mixture risk assessment.

***Comment**: The proposed HI methodologies in the framework are adequately supported for use in preliminary site-specific risk assessments for a specific PFAS mixture composition.*

3. Section 4.4 (Relative Potency Factor; RPF) of the framework document demonstrates the application of a component-based mixture approach, based on dose addition, using available dose-response information (i.e., points-of-departure) from completed EPA human health assessments, and hypothetical exposure information. The example RPFs and corresponding Index Chemical Equivalent Concentration (ICEC) calculations presented are primarily focused on four PFAS with finalized EPA Human Health Assessments: PFOA, PFOS, PFBS, and HFPO dimer acid and GenX chemicals.

A. Please provide specific feedback on whether the RPF approach is a reasonable methodology for estimating risk associated with mixtures of PFAS. If not, please provide an alternative.

***Comment**: The proposed RPF approach is a reasonable methodology for estimating risk associated with specific mixtures of PFAS and is consistent with EPA practice.*

B. Please provide specific feedback on whether the proposed RPF methodology in the framework is scientifically supported for PFAS mixture risk assessment.

***Comment**: The proposed RPF methodology is scientifically supported as an approach for a specific PFAS mixture during a site- or source-specific assessment, but it should be made clear that it is not well suited for the development of general standards that are intended to be applied across sites or sources with different mixing-ratios of component chemicals.*

4. Section 4.5 (Mixture BMD) of the framework document demonstrates the application of a component-based mixture approach using established EPA dose-response modeling (i.e., benchmark dose; BMD) of hypothetical PFAS dose-response data, and hypothetical exposure information.

   A. Please provide specific feedback on whether the Mixture BMD approach is a reasonable methodology for estimating what is in essence a mixture-based point-of-departure. If not, please provide an alternative.

   ***Comment**: The Mixture BMD approach is a reasonable methodology for estimating what is in essence a point-of-departure for a specific mixture composition, but the resulting POD should not be applied across mixtures with different compositions.*

   B. Please provide specific feedback on whether the proposed Mixture BMD methodology in the framework is scientifically supported for PFAS mixture risk assessment.

   ***Comment**: The proposed Mixture BMD methodology in the framework is scientifically supported for a PFAS mixture with a specific mixing-ratio of component chemicals, and can be applied during a site- or source-specific assessment. It is important that the EPA make it clear that it should not be applied in the development of a single standard that is intended to be applied across sites or sources with different mixing-ratios of component chemicals.*

ANALYSIS OF CARDIOVASCULAR DISEASE RISK REDUCTION AS A RESULT OF REDUCED PFOA AND PFOS EXPOSURE IN DRINKING WATER

Overall charge: EPA is seeking SAB comment on the extent to which the approach to estimating reductions in CVD risk associated with reductions in exposure to PFOA and PFOS in drinking water is scientifically supported and clearly described.

Charge Questions

1. Section 4.2 presents EPA's meta-analysis for the total cholesterol dose-response function.

   A. Please provide specific feedback on the extent to which the study selection criteria, the identified studies, and the methodological approach of the meta-analysis are complete and capture up to date scientific literature.

Comment: *Although the EPA describes the study selection criteria, there is no apparent integration of study quality criteria in the meta-analysis. Several of the identified studies were judged by reviewers to be of low (or deficient) quality. Separately, there was no information on the risk of bias analysis for five studies (of 14 total studies considered in the meta-analysis) conducted before 2018. Meta-estimates should be derived for studies which were considered higher quality and/or studies that adjusted for certain confounding factors (lipid-lowering medication). In addition, the EPA meta-analysis did not address the associations between PFOA or PFOS and LDL-C, which were discussed in the draft documents. LDL-C is well established as a causal risk factor for CVD; it is recommended that individuals with high LDL-C (and not high TC) take LDL-C lowering medication to manage CVD risk (Grundy et al. 2019). Presumably, EPA excluded LDL-C from the meta-analysis because the ASCVD risk calculator does not include LDL-C as a predictor. The majority of cholesterol in human lipid profiles is LDL-C so the exclusion of LDL-C does not appear to be a fatal flaw. However, the ASCVD risk model is intended solely for patients with LDL-C <190 mg/dL, without ASCVD, and not on LDL-C lowering therapy. In contrast, studies in the general population did not consistently adjust for lipid-lowering medication and did not exclude individuals with LDL-C ≥ 190 mg/dL.*

An international scientific panel (Andersen et al. 2021b) concluded that correlated net absorption or excretion of bile salts and PFAS in the gut enterocytes could give rise to the apparent associations of cholesterol and PFAS in blood observed in epidemiological studies. It has been demonstrated that several bile acid transporters expressed in enterocytes and hepatocytes can also transport PFAS, suggesting that PFAS could be entrained within the enterohepatic recirculation of bile acids. Co-modulation of the kinetics of bile acids and PFAS at these specific transporters by cholesterol has been shown in the rat. Correlated uptake/biliary excretion of PFAS and bile salts could serve as a confounding link between cholesterol homeostasis and PFAS kinetics, leading to an apparent association between Total Cholesterol (TC) and PFAS concentrations in serum. Importantly, if PFAS and cholesterol kinetics were both correlated with a common confounding process (e.g., bile acid recirculation), the fractional change in TC (compared to the average) in a given study would be expected to be the same as the fractional change in PFAS serum (compared to the average) in a same study. Moreover, this expected relationship would hold whether the exposures were at low or high PFAS

concentrations. In fact, this relationship is consistently seen in studies of PFAS exposures, whether the study subjects are exposed to low environmental concentrations or are occupationally exposed to high concentrations. In other words, a similar fractional change in PFAS concentration (rather than the concentration itself) is associated with a fractional change in TC across many orders of magnitude of exposure concentrations. However, cross-sectional epidemiological studies have evaluated the concentration of PFAS in relation to TC concentrations (using cross-sectional study designs). When evaluating the body of evidence, the apparent effect on cholesterol is stronger at lower concentrations than higher concentrations; however, it is biologically implausible that the potency of PFAS on cholesterol homeostasis decreases as PFAS concentration increases. Therefore, it is more likely that the association is not causal, and that bile acid recirculation distorts the association between PFOA or PFOS and TC.

A recent review of epidemiological studies and other scientific evidence published since 2012 (Steenland et al. 2020) discussed confounding by enterohepatic cycling of PFAS and bile acids as a possible explanation for the positive association between PFOA and total cholesterol. Genius et al. (2014) discussed supporting evidence that cholestyramine, a bile acid sequestrant, also reduced PFOS, PFOA, and PFHxS serum concentrations in humans and rats. Epidemiological studies have not shown increased risks of cardiovascular disease in relation to PFOA or PFOS, even among workers with the highest exposures (Steenland et al. 2015; Steenland and Woskie, 2021; Alexander et al, 2014) or community members exposed to PFOA in contaminated drinking water (Winquist and Steenland, 2014).

B. To inform the CVD risk reduction analysis for those ages 40-89 using the ASCVD risk model, EPA used a meta-analysis approach for the total cholesterol dose-response function. Please provide specific feedback on the extent to which this approach is reasonable for this application, or whether using a single dose-response study (e.g. Dong et al., 2019) selected in the analysis of cholesterol impacts in the *Proposed Approaches for Deriving Maximum Contaminant Level Goals for PFOA and PFOS in Drinking Water* would add additional strengths for the CVD risk reduction application.

Comment: *Small increases in HDL cholesterol have also been reported in relation to PFOA and/or PFOS concentrations in blood serum in cross-sectional studies. In addition to a small, non-statistically significant increase in total cholesterol represented in the pooled estimate seen in Figure A-4 of the Serum Cholesterol Dose-Response Function Appendix, the meta-analysis also showed a small, non-statistically significant increase in HDL cholesterol (Figure A-4 in relation to PFOA and Figure A-8 in relation to PFOS). Presumably, a pooled estimate for the total cholesterol dose-response function reduces the random error associated with relying on a dose-response estimate from a single study. Nevertheless, the pooled estimates showed significant heterogeneity in the underlying studies (I^2 values >70% for both the PFOA (Table A-2) and the PFOS meta-analyses (Table A-3)). This also suggests a single dose-response study should not be relied upon. In any case, the meta-analysis did not address the issue of systematic error and it included studies judged by reviewers to be low quality and/or deficient.*

1. Section 5.1 presents EPA's life table approach methodology.

A. Please comment on the extent to which this analysis is scientifically supported and clearly described. To the extent improvements are suggested, please provide specific changes that are implementable in a U.S. national-level benefits analysis with readily available data.

Comment: *The life table approach is clearly described and is a standard life table approach. Because there is substantial uncertainty introduced in using the ASCVD model to predict CVD events in general (see Comment below, Question 2), and substantial uncertainty regarding the risk of CVD in relation to PFOA and PFOS exposure, a simpler approach to the quantifiable impacts of a potential reduction in cholesterol in relation to reductions in PFOA and PFOS exposure is preferred. Instead of attempting to quantify the impacts of avoided CVD events in relation to reduced total cholesterol, the EPA could consider quantifying avoided health care costs associated with treating high LDL-C (for example, avoided prescriptions for cholesterol-lowering medication) or measuring a reduction in the population eligible for treatment with cholesterol-lowering medication. There are uncertainties with this suggested approach as well; however, it avoids the potential amplification of uncertainty associated with the EPA approach.*

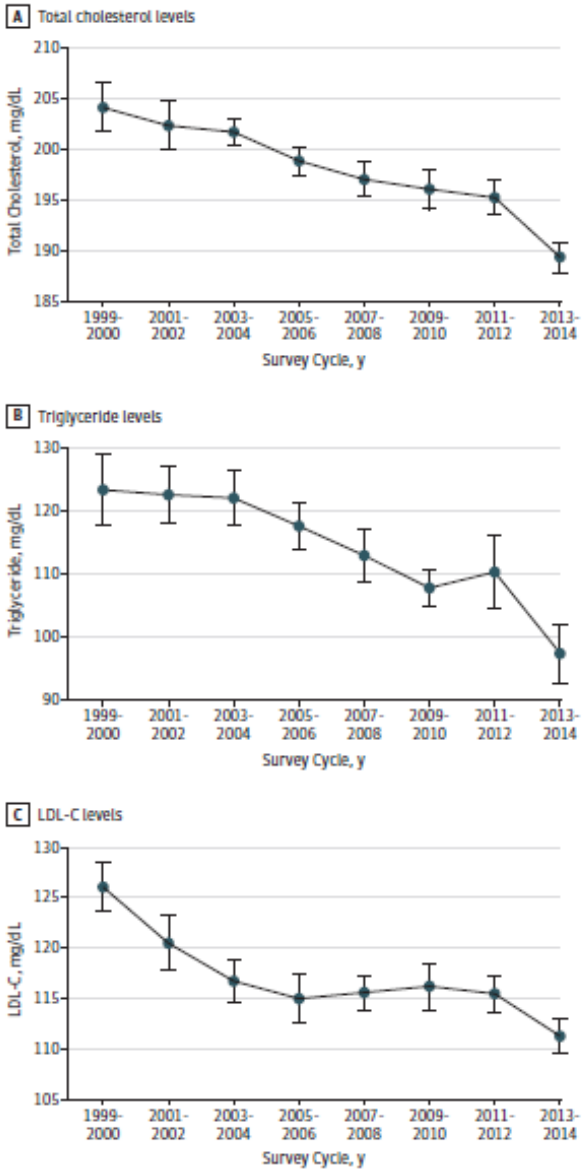
2. Section 5.2 presents EPA's application of the atherosclerotic cardiovascular disease (ASCVD) risk model used to estimate the probability of hard CVD events corresponding to total cholesterol changes.

   A. Please comment on the scientific validity of the ASCVD model application for estimating the probability of first time CVD events in various sub-populations and the extent to which it is clearly described.

Comment: *The ASCVD risk model was developed for use in clinical practice to provide a quantitative risk score for an individual based on an estimate of the 10-year probability of an initial CVD event based on several inputs. The calculator uses age, total and HDL-cholesterol levels, systolic blood pressure, antihypertensive therapy status, history of diabetes and current smoking and assumes that that the individual has LDL-C <190 mg/dL, is free of CVD, and is not taking lipid-lowering medication. The pooled cohort equations that form the basis for the risk estimates are based on older population studies that enrolled volunteers (for example, the Framingham Cohort study, and the Atherosclerosis Risk in Communities (ARIC) study including other cohorts). Although the data may be useful for informing a clinician-patient discussion of managing potential CVD risk for a particular patient, it is not valid for estimating current population-level risks of CVD. Several validation studies have reported that the ASCVD model has overestimated the rate of CVD (DeFilippis et al. 2015; Cook and Ridker, 2014; Rana et al. 2016).The ASCVD model does not account for changes in CVD risk predictors (including cholesterol) over time at a population level. For example, Rosinger et al. (2017) reported that mean cholesterol levels (both TC and LDL-C) and mean triglyceride levels decreased from 1999 to 2014, based on estimates using eight NHANES cycles (see figure below). The decreases over time were similar when stratified by lipid-lowering medication use and are possibly related to the reduction and elimination of artificial trans fats in foods.*

Rosinger et al. 2017:

Figure. Age-Adjusted Total Cholesterol, Triglyceride, and Low-Density Lipoprotein Cholesterol (LDL-C) Trends for US Adults Aged 20 Years and Older, 1999 to 2014



A, Predicted total cholesterol levels and 95% confidence intervals in a sample size of 39 049. B, Predicted log-transformed triglyceride levels and 95% confidence intervals; log-transformed values were exponentiated after the regression, sample size of 17 406. C, Predicted LDL-C levels and 95% confidence intervals in a sample size of 17 096. Figure generated using marginal standardization from age-adjusted linear regression models. Data source: Centers for Disease Control and Prevention/National Center for Health Statistics, the National Health and Nutrition Examination Survey.

SI conversion factors: To convert LDL-C to micromoles per liter, multiply by 0.0259; to convert total cholesterol to micromoles per liter, multiply by 0.0259; to convert triglycerides to micromoles per liter, multiply by 0.0113.

B. Please comment on whether EPA's approach and assumption, of a uniform first CVD event hazard distribution over the 10-year period, is sufficiently robust given current data sources and literature. If additional distributional sources of information are suggested, please provide specific citations/sources for EPA's consideration.

Comment: *Avoided CVD events should not be the basis for the regulatory impact assessment. It is unclear whether the EPA approach and assumption of a uniform first CVD event over a 10-year period is sufficiently robust. Moreover, the link between PFAS exposure and CVD is too tenuous to support a meaningful cost-benefit comparison.*

C. Please comment on the scientific validity of using the ASCVD risk model for estimating reduced CVD risk stemming from changes in total cholesterol in response to reducing exposure to PFOA and PFOS in drinking water.

Comment: *The evidence for a causal relationship between PFOA or PFOS exposure and cardiovascular disease is weak, and it is plausible that PFOA and PFOS exposure is associated with higher total cholesterol levels but without an increased risk of cardiovascular disease (Steenland et al. 2020). It is highly uncertain that reduced PFOA or PFOS exposure will lead to lower cholesterol concentrations that can then be quantified as avoided CVD events. As stated previously, the ASCVD model has not accounted for decreases in total cholesterol over the past 20 years or more and is intended for individuals with LDL-C < 190 mg/dL. A more direct and relevant regulatory cost analysis that requires fewer assumptions (and less uncertainty) is preferred. A regulatory analysis of quantified health risk reduction that focuses on cholesterol reduction (and specifically LDL-C reduction) as the quantified endpoint rather than CVD events avoided would reduce uncertainties, such as use of ASCVD to calculate avoided CVD cases when the epidemiological literature does not show increased risks of CVD in relation to PFOA or PFOS exposure. For example, the EPA could consider avoided use of cholesterol-lowering medication or the reduction in number of adults eligible for treatment for high cholesterol. However, this approach would still not address the possibility that confounding by enterohepatic cycling of PFAS and bile acids is actually responsible for the positive association between PFOA and total cholesterol (Steenland et al. 2020).*

5. Section 7 and Appendix A describe the limitations and uncertainties of the CVD risk reduction analysis. Has EPA clearly described the individual contributions of the sources of uncertainty?

Comment: *If PFOA and PFOS are associated with small increases in TC and small increases in HDL-C, it is biologically plausible that the risks of cardiovascular disease remain unchanged (Steenland et al 2020). The uncertainties described in the ASCVD model included whether PFOA and PFOS potentially impact other risk factors in the ASCVD model (diabetes and systolic blood pressure, for example); however, the ASCVD model does not include all identified or important risk factors (e.g., elevated LDL-C and elevated C-reactive protein (CRP) levels, for example). Although the association between PFOA or PFOS and CRP levels is not well studied, Genser et al. (2015) reported that CRP levels decreased with increasing serum PFOA concentration in an analysis of adults older than 18 years who resided in water districts contaminated with PFOA (the C8 Health Study).*

REFERENCES

Alexander BH, Olsen GW, Burris JM, Mandel JH, Mandel JS. 2003. Mortality of employees of a perfluorooctanesulphonyl fluoride manufacturing facility. Occup Environ Med, 60(10), 722–729.

Andersen ME, Mallick P, Clewell HJ 3rd, Yoon M, Olsen GW, Longnecker MP. 2021a. Using quantitative modeling tools to assess pharmacokinetic bias in epidemiological studies showing associations between biomarkers and health outcomes at low exposures. Environ Res. 197:111183

Andersen ME, Hagenbuch B, Apte U, Corton JC, Fletcher T, Lau C, Roth WL, Staels B, Vega GL, Clewell HJ 3rd, Longnecker MP. 2021b. Why is elevation of serum cholesterol associated with exposure to perfluoroalkyl substances (PFAS) in humans? A workshop report on potential mechanisms. Toxicology. 459:152845.

Andvik C, Jourdain E, Lyche JL, Karoliussen R, Borgå K. 2021. High Levels of Legacy and Emerging Contaminants in Killer Whales (Orcinus orca) from Norway, 2015 to 2017. Environ Toxicol Chem. 2021 Jul;40(7):1850-1860.

Apelberg BJ, Witter FR, Herbstman JB, Calafat AM, Halden RU, Needham LL, Goldman LR. 2007. Cord serum concentrations of perfluorooctane sulfonate (PFOS) and perfluorooctanoate (PFOA) in relation to weight and size at birth. Environ Health Perspect, 115(11), 1670–1676.

Budtz-Jørgensen E, Grandjean P. (2018). Application of benchmark analysis for mixed contaminant exposures: Mutual adjustment of perfluoroalkylate substances associated with immunotoxicity. PLoS ONE 13: e0205388.

Campbell J, Clewell H, Cox T, Dourson M, Ethridge S, Forsberg N, Gadagbui B, Hamade A, Naidu R, Pechack N, Peixe TS, Pruitt R, Prussia A, Rachamalla M, Rhomberg L, Smith J, Verma N. (2022). The conundrum of the PFOA human half-life: An international collaboration. Regulatory Toxicology and Pharmacology. Submitted. Available from: https://tera.org/Alliance%20for%20Risk/Projects/PFOA%20Groups/ARA_2021_PFOA_Summary_December_9.pdf

Checkoway H, Pearce N, Kriebel D. Research Methods in Occupational Epidemiology, Second Edition. Oxford University Press, New York, New York. 2004

Chen MH, Ha EH, Wen TW, Su YN, Lien GW, Chen CY, Chen PC, Hsieh WS. 2012. Perfluorinated compounds in umbilical cord blood and adverse birth outcomes. PLoS One, 7(8), e42474.

Chu C, Zhou Y, Li QQ, Bloom MS, Lin S, Yu YJ, Chen D, Yu HY, Hu LW, Yang BY, Zeng XW, Dong GH. 2020. Are perfluorooctane sulfonate alternatives safer? New insights from a birth cohort study. Environ Int, 135, 105365.

Cook NR, Ridker PM. Further insights into the cardiovascular risk calculator: the role of statins, revascularization, and underascertainment in the Women's Health Study. JAMA Intern Med. 174: 11964-1971.

Dalsager L, Christensen N, Husby S, Kyhl H, Nielsen F, Høst A, Grandjean P, Jensen TK. 2016. Association between prenatal exposure to perfluorinated compounds and symptoms of infections at age 1–4 years among 359 children in the Odense Child Cohort. Environ Int, 96, 58–64.

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(10), 1207–1213.

DeFilippis AP, Young R, Carrubba CJ, McEvoy JW, Budoff MJ, Blumenthal RS, Kronmal RA, McClelland RL, Nasir K, Blaha MJ. (2015). An analysis of calibration and discrimination among multiple cardiovascular risk scores in a modern multiethnic cohort. Ann Intern Med. 2015; 162:266–75.

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.

Grandjean P, Andersen EW, Budtz-Jørgensen E, Nielsen F, Mølbak K, Weihe P, Heilmann C. (2012). Serum vaccine antibody concentrations in children exposed to perfluorinated compounds. JAMA 307: 391-397.

Granum B, Haug LS, Namork E, Stølevik SB, Thomsen C, Aaberge IS, van Loveren H, Løvik 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(4), 373–379.

Grundy SM, Stone NJ, Bailey AL, Beam C, Birtcher KK, Blumenthal RS, Braun LT, de Ferranti S, Faiella-Tommasino J, Forman DE, Goldberg R, Heidenreich PA, Hlatky MA, Jones DW, Lloyd-Jones D, Lopez-Pajares N, Ndumele CE, Orringer CE, Peralta CA, Saseen JJ, Smith SC Jr, Sperling L, Virani SS, Yeboah J. 2018 AHA/ACC/AACVPR/AAPA/ABC/ACPM/ADA/AGS/APhA/ASPC/NLA/PCNA guideline on the management of blood cholesterol: a report of the American College of Cardiology/American Heart Association Task Force on Clinical Practice Guidelines. Circulation. 2019;139:e1082–e1143.

Impinen A, Nygaard UC, Lødrup 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.

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(1), 76–88.

Manzano-Salgado CB, Casas M, Lopez-Espinosa MJ, Ballester F, Iñiguez C, Martinez D, Costa O, Santa-Marina L, Pereda-Pereda E, Schettgen T, Sunyer J, Vrijheid M. 2017. Prenatal exposure to perfluoroalkyl substances and birth outcomes in a Spanish birth cohort. Environ Int, 108, 278–284.

National Research Council. 1992. Biologic Markers in Immunotoxicology. Washington, DC: The National Academies Press. https://doi.org/10.17226/1591.

Office of Environmental Health Hazard Assessment (OEHHA), California Environmental Protection Agency. 2021. FIRST PUBLIC REVIEW DRAFT. Perfluorooctanoic Acid and Perfluorooctane Sulfonic Acid in Drinking Water. July 2021. https://oehha.ca.gov/water/report/perfluorooctanoic-acid-pfoa-and-perfluorooctane-sulfonic-acid-pfos-drinking-water#FirstPubDraft

Okada E, Sasaki S, Saijo Y, Washino N, Miyashita C, Kobayashi S, Konishi K, Ito YM, Ito R, Nakata A, Iwasaki Y, Saito K, Nakazawa H, Kishi R. 2012. Prenatal exposure to perfluorinated chemicals and relationship with allergies and infectious diseases in infants. Environ Res, 112, 118–125.

Olsen GW, Burris JM, Burlew MM, Mandel JH. 2003. Epidemiologic assessment of worker serum perfluorooctanesulfonate (PFOS) and perfluorooctanoate (PFOA) concentrations and medical surveillance examinations. J Occup Environ Med, 45(3), 260–270.

Olsen GW, Zobel LR. 2007. Assessment of lipid, hepatic, and thyroid parameters with serum perfluorooctanoate (PFOA) concentrations in fluorochemical production workers. Int Arch Occup Environ Health, 81(2), 231–246.

Raleigh KK, Alexander BH, Olsen GW, Ramachandran G, Morey SZ, Church TR, Logan PW, Scott LL, Allen EM. 2014 Mortality and cancer incidence in ammonium perfluorooctanoate production workers. Occup Environ Med, 71(7), 500–506.

Rana JS, Tabada GH, Solomon MD, Lo JC, Jaffe MG, Hee Sung S, Ballantyne CM, Go AS. 2016. Accuracy of the Atherosclerotic Cardiovascular Risk Equation in a large contemporary, multiethnic population. J American Coll Cardiol 67:

Rosinger A, Carroll MD, Lacher D, Ogden C. 2017. Trends in total cholesterol, triglycerides, and low-density lipoprotein in US Adults, 1999-2014. JAMA Cardiol. 2: 339-341.

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(3), 386–392.

Savitz DA, Stein CR, Elston B, Wellenius GA, Bartell SM, Shin HM, Vieira VM, Fletcher T. 2012. Relationship of perfluorooctanoic acid exposure to pregnancy outcome based on birth records in the mid-Ohio valley. Environ Health Perspect, 120(8), 1201–1207.

Steenland K, Fletcher T, Stein CR, Bartell SM, Darrow L, Lopez-Espinosa MJ, Ryan PB, Savitz DA. 2020. Review: Evolution of evidence on PFOA and health following the assessments of the C8 Science Panel. Environ Int 145: 106125. Elsevier Ltd. https://doi.org/10.1016/j.envint.2020.106125

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, Zhao L, Winquist A. 2015. A cohort incidence study of workers exposed to perfluorooctanoic acid (PFOA). Occup Environ Med, 72(5), 373–380.

Steenland K, Woskie S. 2012. Cohort mortality study of workers exposed to perfluorooctanoic acid. Am J Epidemiol, 176(10), 909–917.

Stein CR, Savitz DA, Dougan M. 2009. Serum levels of perfluorooctanoic acid and perfluorooctane sulfonate and pregnancy outcome. Am J Epidemiol, 170(7), 837–846.

United Nations Environment Programme; World Health Organization; International Labour Organisation (1996) International Programme on Chemical Safety (IPCS), World Health Organization. 1996. Environmental Health Criteria 180: Principles and Methods for Assessing Direct Immunotoxicity Associated

with Exposure to Chemicals. World Health Organization: Geneva. *https://wedocs.unep.org/20.500.11822/29544*

Weihe P, Kato K. Calafat AM, Nielsen F, Wanigatunga AA, Needham LL, Grandjean P. 2008. Serum concentrations of polyfluoroalkyl compounds in Faroese whale meat consumers. Environ Sci Technol. 2008 Aug 15;42(16): 6291-5.

Winquist A, Steenland K. 2014. Modeled PFOA exposure and coronary artery disease, hypertension, and high cholesterol in community and worker cohorts. Environ Health Perspect, 122(12), 1299–1305.

World Health Organization & International Programme on Chemical Safety. (2012). Guidance for immunotoxicity risk assessment for chemicals. World Health Organization. *https://apps.who.int/iris/handle/10665/330098*

WHO. (2018). WHO Immunological basis for immunization series module 3: tetanus update 2018. Retrieved from http://www.who.int/immunization/documents/ISBN9789241513616/en/

WHO. (2017). Tetanus Vaccines: WHO Position paper – February 2017. Retrieved from https://www.who.int/publications/i/item/tetanus-vaccines-who-position-paper-february-2017

Appendix B

WITAF 056 Technical Memorandum Update: PFAS National Cost Model Report

WITAF 56 TECHNICAL MEMORANDUM UPDATE

PFAS National Cost Model Report

B&V PROJECT NO. 409850

PREPARED FOR

American Water Works Association

American Water Works Association

26 MAY 2023

BLACK & VEATCH

Table of Contents

LIST OF TABLES

LIST OF FIGURES

Abbreviations

AACE	Association for the Advancement of Cost Engineering
AWWA	American Water Works Association
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CFPUA	Cape Fear Public Utility Authority
CIP	Clean-In-Place
CWS	Community Water System
DBP	Disinfection Byproducts
EBCT	Empty Bed Contact Time
EPA	U.S. Environmental Protection Agency
EPTDS	Entry Point to the Distribution System
FRP	Fiberglass Reinforced Plastic
ft	Feet
GAC	Granular Activated Carbon
g/cc	Grams per Cubic Centimeter
gpm	Gallons per Minute
gpm/sf	Gallons per Minute per Square Feet
HRT	Hydraulic Retention Time
IX	Ion Exchange
kWh	Kilowatt-hour
lb/gal	Pounds per Gallon
LHHCWD	La Habra Heights County Water District
MCL	Maximum Contaminant Level
mgd	Million Gallons per Day
mg/L	Milligrams per Liter
NF	Nanofiltration
NTNCWS	Non-Transient Non-Community Water System
NPDWR	National Primary Drinking Water Regulation
PFAS	Per- And Polyfluoroalkyl Substances
PFBS	Perfluorobutane Sulfonic Acid
PFHpA	Perfluoroheptanoic Acid
PFHxS	Perfluorohexane Sulfonate
PFNA	Perfluorononanoic Acid
PFOA	Perfluorooctanoic Acid
PFOS	Perfluorooctane Sulfonic Acid
ppt	Parts Per Trillion
PWS	Public Water System
RO	Reverse Osmosis
SDWIS	Safe Drinking Water Information System
sf	Square Feet
SLR	Surface Loading Rate
T&O	Taste and Odor

TDH	Total Dynamic Head
TDS	Total Dissolved Solids
TOC	Total Organic Carbon
UCMR	Unregulated Contaminant Monitoring Rule
WITAF	Water Industry Technical Action Fund

1.0 Acknowledgement

This study was a collaborative effort; many individuals and utilities spent time compiling data, answering questions, and making contacts. We wish to thank the following utilities for sharing data and Steering Committee Members for their time and insight:

Partner Utilities:

- Cape Fear Public Utility Authority (CFPUA)
- City of Ann Arbor
- Greater Cincinnati Water Works
- Plainfield Charter Township
- City of North Miami Beach
- Miami-Dade County Water and Sewer Department
- Tucson Water

Steering Committee Members:

- Amy Stoffer – Northern Kentucky Water District
- Cynthia Lane – Platte Canyon
- Carel Vandermeyden – CFPUA
- Robert Cheng – Coachella Valley Water District
- Zaid Chowdhury – Garver USA
- Chuck Hertz – Retired

2.0 Introduction

Known as “forever chemicals” because they do not easily biodegrade, per- and polyfluoroalkyl substances (PFAS) are drawing increased scrutiny from health agencies, water utilities, and the public for their presence in drinking water and their effects on human and environmental health. They have quickly become contaminants of great concern in drinking water.

Six PFAS compounds were monitored in finished drinking water as part of the Third Unregulated Contaminant Monitoring Rule (UCMR 3) between 2013 and 2015 to quantify their prevalence across the United States. The UCMR program provides the U.S. Environmental Protection Agency (EPA) with nationally representative occurrence data to inform drinking water regulations. Using the results from UCMR 3, in February 2021, the EPA published a final determination to regulate perfluorooctanoic acid (PFOA) and perfluorooctane sulfonic acid (PFOS) and signaled an interest in considering the regulation of additional PFAS. On March 14, 2023 the EPA announced the first proposed National Primary Drinking Water Regulation (NPDWR) for six PFAS compounds, including PFOA, PFOS, perfluorononanoic acid (PFNA), hexafluoropropylene oxide dimer acid (HFPO-DA, commonly known as GenX Chemicals), perfluorohexane sulfonic acid (PFHxS), and perfluorobutane sulfonic acid (PFBS). The deadline for public comment on this proposed regulation is May 30th, 2023 and the EPA has publicly committed to promulgate the PFAS NPDWR by the end of 2023.

U.S. federal laws and executive orders stipulate that the U.S. EPA estimate the cost of compliance for this new primary drinking water regulation. Black & Veatch was selected by the American Water Works Association (AWWA) to develop a national cost estimate for water systems to remove PFAS from drinking water to better understand the financial impacts to communities and the costs to comply with a national primary drinking water regulation, a policy that could impact each of the more than 66,500 public water systems.

The project was funded by the Water Industry Technical Action Fund (WITAF), which is managed by the AWWA’s Water Utility Council to support projects, studies, analyses, reports, and presentations in support of the organization’s legislative and regulatory agenda. The national cost estimate and its cost models, developed under WITAF 056, are intended to support to AWWA’s engagement with the U.S. EPA and Congress on the differences in financial impacts of treating drinking water to various PFAS regulatory limits. WITAF funded a separate project (WITAF 057) to generate a national PFAS occurrence database using data from state monitoring and UCMR3. This national database was used as an input for the WITAF 057 project.

The national cost modeling tool programmatically evaluates each public water system (PWS) with occurrence data from WITAF 057 to generate a dataset of the most probable capital and operating costs. Those costs are then scaled up nationally to account for the PWSs without data captured in WITAF 057 to quantify the national cost of compliance of a proposed regulation, bringing flexibility for data-driven responses to EPA cost assessments. This project brought together occurrence data, cost data, and best practice design methodology to help ensure the U.S. EPA’s proposed national primary drinking water regulations for PFAS accurately reflect cost estimates for drinking water treatment.

3.0 PFAS Treatment Technologies

Treatment strategies for PFAS in drinking water include proven, commercially available technologies as well as emerging technologies. Many of these developing technologies have been demonstrated on the bench scale but have not yet been proven at the full scale or are not yet commercially available. Commercially available technologies that have been demonstrated at full scale in the field to reduce concentrations of PFAS in drinking water are limited to the following:

- Granular activated carbon (GAC).
- Ion exchange (IX).
- Nanofiltration (NF) and reverse osmosis (RO).

Treatment considerations for the application of each of these technologies are described in the following subsections.

3.1 Granular Activated Carbon

GAC media is a well-known adsorbent for organics and has been widely applied in water treatment. GAC is produced from carbon-based materials such as coal, coconut shells, peat, or wood that has been "activated" to produce a highly porous media with adsorptive properties. The pores contain sites on which organic compounds become attached and are adsorbed onto the activated carbon matrix.

GAC treatment applications include removal of organics, such as color, disinfection byproducts (DBP) and their precursors, taste and odor (T&O) causing compounds, industrial chemicals, and emerging contaminants such as PFAS, endocrine disrupting compounds, and pharmaceuticals and personal care products. Each of these contaminants compete for adsorption sites on GAC media with targeted PFAS if present. In some cases, co-adsorption can be viewed as a benefit for using GAC as the co-contaminants are simultaneously removed. Cost analyses and removal performance models must balance competitive adsorption of co-contaminants and its associated detrimental performance impact on PFAS removal.

GAC has a finite capacity for adsorbing compounds. High concentrations of organics or high flow rates will lead to more frequent media replacement. In general, short-chained PFAS are less readily adsorbed and less strongly bound than long chain compounds. The overall efficacy of GAC removal of PFAS highly dependent on the water matrix, the water treatment goals, and the design of the system. One of the most important design parameters is the empty bed contact time (EBCT), or the time during which the water is in contact with the media bed (also the duration at which adsorption can occur), assuming the water flows through the entire bed at a constant velocity. A desired EBCT will result in breakthrough when the adsorptive capacity of the media has been exhausted. The media must be either replaced or reactivated at that time.

3.1.1 Implementation and Operational Considerations

GAC applied for PFAS removal is most effective when used solely as an adsorbent. Conventional granular media filters containing GAC are typically designed for short EBCTs and must be frequently backwashed for removal of particulate material that is retained in the media. Such backwashing disrupts the adsorption front. Short EBCTs and backwashing lead to fast breakthrough of contaminants and underutilization of GAC media. If a water treatment facility contains conventional filters, contactors for GAC adsorption are typically located downstream.

Process selection (including GAC media selection) is typically confirmed through demonstration testing (bench-, pilot- or full-scale studies) to account for the unique characteristics of the source water.

GAC adsorption treatment systems installed for PFAS removal typically provide a 10 to 20 minute EBCT and a surface loading rate of 4 to 10 gallon per minute (gpm) per square foot of media (gpm/sf). PFAS adsorbers are applied in two main configurations: pressure vessels or gravity basins.

- Pressure vessel configurations are more common in small systems (less than approximately 10 million gallons per day [mgd]). Pre-engineered pressure-vessel type GAC treatment systems are widely available. Vessels are typically carbon steel or fiberglass reinforced plastic (FRP). Pressure vessels may be installed in single (parallel) or dual stage (series/lead-lag) arrangements.
    - The single stage arrangement allows for columns to be operated in various stages of breakthrough or exhaustion, resulting in an overall effluent below the treatment target. This arrangement can result in better media utilization, produce a more consistent product water quality, and lessen impact of potential overruns on individual vessels. Single stage systems typically include N+1 redundancy.
    - The dual stage arrangement allows for simultaneous production during media replacement, and sampling between vessels ensures that lag vessel effluent always meets treatment targets. The lead vessel can be in service until the media is completely exhausted, leading to higher utilization of the adsorbent media. The dual stage arrangement includes built-in redundancy as either the lead or lag vessel can be removed from service without reducing the treatment flow rate. Thus, no dedicated redundant vessels are typically provided.
- To avoid an excessive number of pressure vessels, gravity basin configurations are typically applied by large systems with design flows greater than approximately 10 mgd. Gravity basins are typically single stage and operated at various stages of breakthrough, similar to a single stage pressure vessel arrangement. The basins themselves are typically constructed of concrete with an N+1 redundancy because of the single stage arrangement.

Exhausted GAC filter media will be saturated with PFAS. Bulk GAC can be reactivated by the media supplier through thermal treatment at high temperatures (up to 1800˚ F) to remove and destroy adsorbed contaminants (Rebecca DiStefano, 2022). This reactivation process restores the media's adsorptive capacity, allowing the media to be returned for reuse. GAC is sometimes regenerated by heating the media to temperatures typically less than 400˚ F to remove a portion of the adsorbed contaminants. However, this process will not remove all the compounds and will not destroy the PFAS compounds; therefore, it is not appropriate for GAC utilized for PFAS removal. Media suppliers may not accept the low volumes of GAC required by small systems for reactivation, forcing them to dispose of spent GAC and replace it with new (virgin) material.

Disposal alternatives for exhausted GAC that will not be reactivated for municipal reuse include disposal by reactivation for industrial reuse, incineration, and landfilling. The cost of each disposal method depends on proximity to disposal sites, hazardous waste classification, and volume of material. Disposal costs can be a significant operational cost for GAC treatment systems.

The EPA proposed to designate PFOS and PFOA as hazardous substances under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) in August of 2022. This designation is expected to limit the disposal sites willing to accept spent GAC media. Additionally, the practice of reactivating GAC media contaminated with PFAS is expected to be more limited in drinking water applications.

3.1.2 Assumptions for Cost Estimation

The cost model includes capital costs, annual operating and maintenance costs, life-cycle costs, and annualized costs. The assumptions that drove the results of those cost estimates are summarized in this section.

The costs for GAC contactors depend on the contactor type, size, number, and ancillary processes such as backwash pumps/recovery basins and contactor influent pumps/wet wells. The primary process design assumptions for each of these factors are summarized in Table 3-1.

Table 3-1 GAC Design Process Assumptions

Contactor Type	Parameter		Assumption/Input
Pressure Vessel	Treatment Plant Capacity		1-12 mgd
	Surface Loading Rate(Note 1)		4-10 gpm/sf (most likely 6 gpm/sf)
	Empty Bed Contact Time(Note 1)		10-20 min (most likely 18 min)
	Vessel Diameter		6-12 ft
	Arrangement		Dual Stage
	Redundancy		None
	Influent Pump Station	TDH (total dynamic head)	45 ft
		Design HRT (hydraulic retention time)	15 min
Gravity Basin	Treatment Plant Capacity		> 12 mgd
	Surface Loading Rate(Note 1)		4-10 gpm/sf (most likely 4 gpm/sf)
	Empty Bed Contact Time(Note 1)		10-20 min (most likely 18 min)
	Filter Dimensions		8-20 ft cell width, 2:1 length to width ratio
	Arrangement		Single Stage
	Redundancy		N+1
	Influent Pump Station TDH, design HRT		30 ft, 15 min

Contactor Type	Parameter		Assumption/Input
Common	Backwash[Note 2]	Loading Rate	13 gpm/sf
		Duration	30 min
		Frequency	30 days
		Pump Design TDH	60 ft
	Influent Pump Station[Note 3]	Pump Efficiency	70%
		Motor Efficiency	85%
	Backwash Water Recovery Basin[Note 4]	Water Depth	20 ft
		Backwash Cycles Held	1.0
	GAC Media	Apparent Density	0.5 gram per cubic centimeter (g/cc)
	Contactor Area Factor (for pipe gallery and appurtenances)		2.0

Notes:

1. For adsorptive media, the major specified process design inputs are the surface loading rate (SLR) and the EBCT. For each of these factors, a minimum, maximum, and most likely number was assumed using feedback from existing treatment systems. The minimum, maximum, and most likely numbers for the published model outputs are summarized herein. National variability in SLR and EBCT is included in the model using a Monte Carlo simulation. The details of how this statistical method was employed within the cost modeling tool is described in Section 5.3.
2. Backwash pumps are required for periodic backwashing of the media.
3. An influent pump station is presumed to be required to accommodate the additional headloss necessary to an existing process train.
4. Backwash recovery basin omitted from systems for size category 1 and 2.

3.2 Ion Exchange

IX is an adsorptive water treatment process that involves the selective exchange of ions in solution with ions bound to a resin matrix. IX has a long history in water treatment, and resins are manufactured for a variety of contaminants, including PFAS. Several manufacturers provide specific IX resins designed to be selective for PFAS as the market has expanded for their use. Some resins originally intended for removal of other contaminants (such as perchlorate) have shown a high degree of selectivity and capacity for PFAS as well.

IX resins, like GAC, have a limited capacity for adsorption. The adsorptive capacity of IX resins is affected by contaminant concentrations and flow rates in the same manner as GAC. However, the IX resins surveyed have proven to be highly selective toward PFAS removal, exhibiting minimal removal of other contaminants. This may result in a greater adsorptive capacity for PFAS compared to GAC, without, however, the co-contaminant removal benefits of other technologies. In general, short-chained PFAS are less readily adsorbed and less strongly bound than long chain compounds. The overall efficacy of IX for PFAS removal is highly individual to the water matrix, the water treatment goals, and the design of the system.

An IX treatment process does not result in a fixed percentage removal of a contaminant over time, as there is a variable degree of contaminant removal and gradual or sharp contaminant breakthrough. Although it is selective to certain contaminant groups, the resin can experience interference from other compounds in the water matrix. The most preferred compound will tend to exhibit long runs and sharp breakthroughs; less preferred compounds will have earlier, more gradual breakthroughs.

Exhaustion of the media is determined (in a fashion similar to that for GAC) through the measure of the contaminant in the effluent (breakthrough). When the adsorptive capacity has been exhausted, the resins require replacement or regeneration. Because of the proposed CERCLA hazardous substance designations for PFOA and PFOS as discussed in Subsection 3.1.1, single use (fixed-bed) systems are currently being considered for IX, requiring disposal of spent media and replacement with new resin when exhausted. PFAS destruction technologies are currently in research and development that may be able to destroy PFAS in the brine stream, although that technology is not yet matured enough for full-scale implementation.

Fixed-bed IX has been demonstrated at full scale in the field as a proven PFAS removal technology. Fixed-bed ion exchangers applied for PFAS removal consist of carbon steel or FRP pressure vessels and typically 1.5 to 3 minutes of EBCT (as compared to 10 to 20 minutes for GAC). IX can be favorable because of the smaller footprint required.

3.2.1 Implementation and Operational Considerations

The efficacy of an IX treatment system will likely be improved by a pretreatment step to remove interferences such as suspended solids, particulate natural organic matter, and colloidal compounds. Commercially available filters can be selected depending on the pretreatment needs to improve the treatment capacity of the IX system. This prefiltration step can prevent deposition of fine particles on the resin, reduce pressure drop across a column, and increase run time.

Process selection (including resin selection) is typically confirmed through demonstration testing (bench-, pilot- or full-scale studies) to account for the unique characteristics of the source water.

Ion exchange treatment systems are conventionally installed in pressure filters in lieu of gravity basins. As with GAC, the pressure vessels can be implemented in single or dual stage arrangements. Considerations for the single or dual stage arrangements are summarized in Subsection 3.1.1.

Exhausted IX resin will be saturated with PFAS. Disposal alternatives for exhausted IX resins include incineration and landfilling. The cost of each disposal method depends on proximity to disposal sites, hazardous waste classification, and volume of material. Disposal costs can be a significant operational cost for IX treatment systems.

3.2.2 Assumptions for Cost Estimation

The costs for IX Contactors depend on the contactor type, size, number, and ancillary processes such as backwash pumps/recovery basins and contactor influent pumps/wetwells. The primary process design assumptions for each of these factors are summarized in Table 3-2.

Table 3-2 IX Design Process Assumptions

Parameter		Assumption/Input
Surface Loading Rate[Note 1]		5-12 gpm/sf (most likely 8 gpm/sf)
Empty Bed Contact Time[Note 1]		1.5-3.0 min (most likely 2.0 min)
Vessel Diameter		4-12 ft
Contact Mode		Lead-Lag
Redundancy		None
Influent Pump Station [Note 2]	Pump Efficiency	70%
	Motor Efficiency	85%
	TDH	60 ft
	Design HRT	15 min
Backwash [Note 3]	Loading Rate	5 gpm/sf
	Duration	30 min
	Frequency	30 days
	Pump Design TDH	60 ft
Backwash Water Recovery Basin [Note 4]	Water Depth	20 ft
	Backwash Cycles Held	1.0
IX Resin	Apparent Density	1.05 g/cc
Contactor Area Factor (for pipe gallery and appurtenances)		2.0

Notes:

1. For adsorptive media, the major specified process design inputs are the SLR and the EBCT. For each of these factors, a minimum, maximum, and most likely number was assumed using feedback from existing systems. The minimum, maximum, and most likely numbers used for the published model outputs are summarized herein. National variability in SLR and EBCT is included in the model using a Monte Carlo simulation. The details of how this statistical method was employed within the cost modeling tool is described in Section 5.3.
2. An influent pump station is presumed to be required to accommodate the additional headloss necessary to an existing process train.
3. Backwash pumps are required for periodic backwashing of the media.
4. Backwash recovery basin omitted from systems for size category 1 and 2.

3.3 Reverse Osmosis and Nanofiltration

RO and NF are membrane-based water treatment processes in which a semi-permeable barrier removes dissolved contaminants from water. This capability is attractive when considering the need to remove total dissolved solids (TDS), specific ions such as calcium, magnesium, sodium, chloride, sulfate, and hardness; DBP precursors; and T&O causing compounds as well as high levels of PFAS. RO/NF processes are commonly applied in water treatment plants and have applications ranging from desalination of brackish water, softening, and the removal of nitrate, agricultural chemicals (e.g., atrazine), color, total organic carbon (TOC), DBP precursors, and PFAS. Both RO and NF processes are capable of a high rejection of PFAS. While RO/NF systems are more expensive than GAC or IX systems, they are most viable when the GAC/IX replacement frequency requirements are cost-prohibitive because of high concentrations of influent PFAS.

The key differences between RO and NF are salt passage and feed pressure. RO membranes reject a higher percentage of dissolved ions in the feed water and require a greater feed pressure than NF membranes. NF membranes preferentially remove larger divalent ions or molecules compared to monovalent ions. Thus, NF systems generally exhibit lower energy use and lower operating cost than RO systems. The lower feed pressure required for NF generally translates to a slightly favorable capital cost in relation to RO systems treating the same flow rate. However, the benefits of higher salt rejection and flexibility of systems designed for RO to utilize either NF or RO membranes typically results in utilities favoring RO over marginally lower cost NF systems.

For a typical RO/NF system, membrane elements are mounted into pressure vessels that are arranged in stages, banks, or arrays. The number of stages required depends on specified recovery. Two stages are typically used for recovery less than 80 percent, and three stages are required for higher recovery. RO/NF is a cross flow filtration method, in which only a portion of the feedwater becomes permeate (finished water). The remainder leaves the system as concentrate (brine) that carries away the concentrated material before precipitation or scaling forms on the membrane surface or in the device. Antiscalant is used to control the precipitation of sparingly soluble salts such as calcium carbonate, calcium sulphate, barium sulfate, calcium fluoride, silicon dioxide, etc.

3.3.1 Implementation and Operational Considerations

The recovery of the RO/NF treatment systems depends on the concentrations of the sparingly soluble salts and typically ranges from 75 to 85 percent. Pretreatment requirements include pH depression, antiscalant chemical products to reduce scaling, and cartridge filters to protect the RO/NF membranes from particulates.

The combination of pH depression in the feedwater and the removal of alkalinity through the process results in a low pH (acidic) finished water. Gases pass through NF/RO membranes, resulting in the potential need for removal of hydrogen sulfide and carbon dioxide from the treated water. Post-treatment generally consists of gas stripping through a decarbonation tower and chemical conditioning by addition of a base such as lime or sodium hydroxide (caustic) to raise pH, alkalinity, and hardness to render the water less corrosive. Sometimes a corrosion inhibitor is also added to prevent distribution system corrosion.

A major challenge to implementing centralized NF/RO treatment for PFAS removal is in dealing with the concentrated waste stream generated by the treatment process. Contaminants are rejected into a waste brine stream that is typically around 15 percent by volume of the feedwater (for low salinity feed waters) and 4 to 7 times more concentrated than the raw water fed to the membranes. As a result,

additional raw water is required to achieve the desired finished water capacity, and the waste stream requires disposal. Traditional alternatives for disposal include sending the stream to a downstream water reclamation facility, discharging to surface water, or injection into underground deep wells. However, because of the CERCLA regulations for PFOA and PFOS as discussed in Subsection 3.1.1 and pending effluent limit goals for PFAS, concentrate treatment may be required before disposal using these methods.

3.3.2 Assumptions for Cost Estimation

The costs for RO systems depend on the number of trains, permeate flow, and ancillary processes such as the RO feed tank, low-pressure feed pump, high-pressure feed pump, chemical pretreatment, chemical post-treatment, flush pump/tank, clean-in-place (CIP) system, decarbonation system, building requirements, and brine disposal. The primary process design assumptions for each of these factors are summarized in Table 3-3.

Table 3-3 RO Design Process Assumptions

Sub-System	Parameter	Assumption/Input
RO System Design	Recovery(Note 1)	70-85%
	PFAS Rejection	95%
	RO Element Membrane Area	400 ft^2
	Design Flux	15 gallons per foot-squared per day (gfd)
	Redundancy	N+1
	Concentrate Recycle	0%
	Number of Elements per Pressure Vessel	6
	First Stage Pressure Vessel Ratio	4
	Second Stage Pressure Vessel Ratio	2
	Third Stage Pressure Vessel Ratio	1
RO Feed Tank	Hydraulic Detention Time	30 min
RO Low Pressure Feed Pump Sizing	Pump Design TDH	30 ft
RO High Pressure Feed Pump Sizing	Pump Design TDH	350 ft
Chemical Pretreatment(Note 2) Antiscalant Chemical	Density	10.01 pounds per gallon (lb/gal)
	Design Dose	3 mg/L
	Storage	30 days

Sub-System	Parameter	Assumption/Input
Chemical Pretreatment[(Note 2)] Sulfuric Acid (98%)	Density	15.26 lb/gal
	Design Dose	30 milligrams per liter (mg/L)
	Storage	30 days
Chemical Post-Treatment[(Note 2)] Caustic (50%) or Liquid Lime	Density	12.78 lb/gal
	Design Dose	45 mg/L
	Storage	30 days
RO Flush Pump Sizing	Flow Rate per Pressure Vessel	30 gpm/1st stage pressure vessel
	Pump Design TDH	140 ft
	Flush Frequency	12 hrs/yr/train
RO Flush Tank Sizing	Volume per Pressure Vessel	7 cubic feet (ft^3)
	Number of Flushes in Tank	2
	Safety Factor	50%
CIP System Sizing	Flush Flow	50 gpm/1st stage pressure vessel
	Time/skid	4 minutes
	CIP Pump TDH	140 ft
	CIP Interval	90 days
	Time/CIP	6 hrs
	CIP Temperature Increase	65 °F
	Heater Losses	10%
Forced Draft Degasifier (decarbonation)	Loading Rate	30 gallons per minute per square foot ($gal/min/ft^2$)
Building Calculations	RO Equipment Area Factor	2.0
	Unit Area	880 ft^2/mgd

Sub-System	Parameter	Assumption/Input
Brine Disposal		Deep Well Injection
Deep Well Injection	Flow per well	1 mgd

Notes:

1. For RO, the critical design input is percent recovery. A minimum and maximum recovery, but no most likely number, is specified. The minimum and maximum recovery used for the published model outputs are summarized herein. National variability in recovery is included in the model using a Monte Carlo simulation. The details of how this statistical method was employed within the cost modeling tool is described in Section 5.3.
2. Chemical systems include pumps, bulk storage, piping, and containment. No day tanks were included in the estimate.

4.0 Estimating National Occurrence

To estimate the costs of removing PFAS from drinking water nationally, national occurrence must be characterized. In parallel to this project, AWWA funded WITAF 057 to compile an occurrence database for PFAS in drinking water. In addition to data available for UCMR 3, WITAF 057 facilitated the collection of PFAS monitoring data from state databases and integrated these sources into a single data set. PWSs in this database included only active Community Water Systems (CWSs) and active Non-Transient Non-Community Water Systems (NTNCWSs). The inactive and transient non community water systems were eliminated from the dataset. Consecutive systems receiving all water from treated water wholesaler systems were not excluded from the database or from representation in the national cost estimation.

The WITAF 057 dataset consisted of 7,842 PWSs within these categories as compared to the 49,193 PWSs in the Safe Drinking Water Information System (SDWIS). To account for this incomplete occurrence data, the percent of systems impacted by a potential PFAS regulation within each system size category was multiplied by the active number of CWSs or NTNCWSs in EPA's SDWIS system at each size category to estimate the anticipated number of total water systems impacted in each size category. This methodology therefore assumed that existing occurrence data is representative of national occurrence. This assumption is considered conservative given a significant fraction of existing occurrence data came from UCMR 3, where the reporting limits of 20 parts per trillion (ppt) and 40 ppt for PFOA and PFOS, respectively, likely bias existing occurrence data to underrepresent true national occurrence that would be measured using the current reporting limits.

Monitoring data for PFAS compounds in the WITAF 057 database included more than 30 individual compounds but for this work was limited to the six PFAS covered by UCMR 3: PFOS, perfluoroheptanoic acid (PFHpA), PFHxS, PFNA, and PFBS. As compiled, the WITAF 057 database includes all monitoring results under UCMR 3 and various state monitoring programs, which may include multiple sample results for specific PFAS at a given PWS. Reported data were reviewed to ensure correct translation of reporting units; fields were included for PWS identification number, state, number of people served, source type, and system type. These data were analyzed to determine the maximum and average sample results for each PFAS at each PWS in the database.

5.0 Individual Treatment Facility Cost Methodology

The next step in estimating the national costs to remove PFAS from drinking water is to use the occurrence database to estimate the costs associated with treatment for individual PWSs. The following subsections summarize how capital, operating, and life-cycle costs are calculated for each system and for each technology.

The spreadsheet tool developed to perform this task accepts inputs for individual or combined target effluent levels for the six PFAS compounds represented in the database. After both occurrence data and potential regulatory levels are input, Visual Basic scripts within Excel may be initiated by a user to run a Monte Carlo analysis and generate a 10th percentile, 90th percentile, and most probable costs for the capital, operations and maintenance (O&M), and life-cycle costs for a typical entry point to the distribution system (EPTDS) for each PWS in the database. For each system, the tool selects the treatment technology with the lowest life-cycle cost.

This methodology assumes installation of a treatment system at each EPTDS associated with PWSIDs where the maximum PFAS concentration is greater than the potential regulatory level for the corresponding PFAS. The details of individual system and EPTDS cost methodology are described in the following subsections. A list of output fields generated by the cost modeling tool for each PWS with occurrence data is shown in Table 5-1.

Table 5-1 Model Outputs for Individual PWS with Occurrence Data

Model Outputs for Each PWS with Occurrence Data
Design Flow (mgd)
Average Flow (mgd)
Capital Expenditure for GAC Vessels
Annual Operations and Maintenance Costs for GAC Vessels
Life-Cycle Costs for GAC Vessels
Capital Expenditure for GAC Basins
Annual Operations and Maintenance Costs for GAC Basins
Life-Cycle Costs for GAC Basins
Capital Expenditure for Ion Exchange Vessels
Annual Operations and Maintenance Costs for Ion Exchange Vessels
Life-Cycle Costs for Ion Exchange Vessels
Capital Expenditure for Reverse Osmosis
Annual Operations and Maintenance Costs for Reverse Osmosis
Life-Cycle Costs for Reverse Osmosis
Capital Expenditure for Lowest Life-Cycle Cost Technology
Annual Operations and Maintenance Costs for Lowest Life-Cycle Cost Technology

Model Outputs for Each PWS with Occurrence Data
Life-Cycle Costs for Lowest Life-Cycle Cost Technology
10th Percentile Capital Expenditure for Lowest Life-Cycle Cost Technology
10th Percentile Operations and Maintenance Cost for Lowest Life-Cycle Cost Technology
10th Percentile Life-Cycle Cost for Lowest Life-Cycle Cost Technology
90th Percentile Capital Expenditure for Lowest Life-Cycle Cost Technology
90th Percentile Operations and Maintenance Cost for Lowest Life-Cycle Cost Technology
90th Percentile Life-Cycle Cost for Lowest Life-Cycle Cost Technology
Capital Expenditure for Manganese Pretreatment
Annual Operations and Maintenance Costs for Manganese Pretreatment
Life-Cycle Cost for Manganese Pretreatment
Lowest Life-Cycle Cost Treatment Technology

5.1 Determining Design Parameters

5.1.1 Treatment Design Flow Determination

PWS data available in SDWIS do not include water usage data for each PWS and EPTDS. Instead, 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. While not reflective of each state's dynamics with respect to water usage, this was considered a reasonable number from a national perspective. Peaking factors for different size systems from the EPA's "Cost and Technology Document for Final Groundwater Rule" were used and are shown in Table 5-2. The trend of this dataset was best fit to a power equation to calculate peaking factor as a function of average daily flow as shown on Figure 5-1.

Table 5-2- EPA Peaking Factor for Various Average System Flows

Design Flow (MGD)	Average Flow (MGD)	Peaking Factor	Design Flow (MGD)	Average Flow (MGD)	Peaking Factor
0.007	0.0015	4.7	2	0.77	2.6
0.022	0.0054	4.1	3.5	1.4	2.5
0.037	0.0095	3.9	7	3	2.3
0.091	0.025	3.6	17	7.8	2.2
0.18	0.054	3.3	22	11	2
0.27	0.084	3.2	76	38	2
0.36	0.11	3.3	210	120	1.8
0.68	0.23	3	430	270	1.6
1	0.3	3.3	520	350	1.5

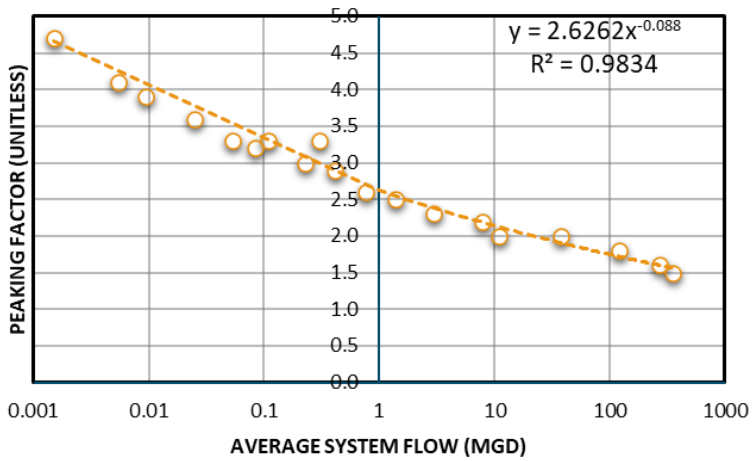


Figure 5-1 Peaking Factor as a Function of Average System Flow

The treatment design flow per EPTDS was determined by Equation 1:

$$Design\ Flow\ per\ EPTDS = \frac{(number\ of\ customers\ per\ PWS)(150\ gpdc)(peaking\ factor)}{EPTDS\ per\ PWS\ size\ category} \quad (1)$$

Where:

$$Peaking\ factor = 2.6262(PWS\ Average\ Flow)^{-0.088}$$

The estimated number of EPTDS per system size is based on an evaluation by EPA published with the proposed national primary drinking water regulation for PFAS from March 2023. The number of EPTDS per system broken out by groundwater and surface water systems within each system size bin is summarized in Table 5-3.

Table 5-3 Number of EPTDS as a Function of System Size

Size Category	Population Range	Entry Points/System	
		Groundwater	Surface Water
1	0-100	1	1
2	101-500	1	1
3	501-1,000	2	1
4	1,001-3,300	2	1
5	3,301-10,000	2	1

Size Category	Population Range	Entry Points/System	
		Groundwater	Surface Water
6	10,001-50,000	4	1
7	50,001-100,000	10	2
8	100,001-1,000,000	12	2
9	>1,000,001	39	4

5.1.2 Water Quality Considerations Incorporated

5.1.2.1 Influent and Effluent PFAS Levels

For each PWS in the occurrence database, any single PFAS monitoring result above either existing state or potential regulatory limit was assumed to incur a capital expenditure for treatment. Data down to the resolution of each individual source was not considered for this modeling effort; instead, the number of projected water treatment facilities per system was based on the EPTDS factors as summarized in the previous section. Maximum PFAS monitoring data were assumed to compel treatment for the PWS as a whole and, thus, all the projected water treatment facilities. The average PFAS monitoring data were used to estimate long-term costs of removal (annual O&M costs).

The target effluent PFAS levels for treatment was determined as an input percentage of a potential regulatory limit. For example, treatment could be triggered at 80, 90, or 100 percent of the potential regulatory level. For this work, a threshold of 80 percent was used in alignment with previous practice for estimating costs of potential regulations for drinking water, since water systems will target and operate below this threshold to ensure that the limit is not exceeded if the water quality suddenly increases.

5.1.2.2 Other Water Quality Considerations

Other water quality contaminants may impact PFAS treatment performance (and therefore costs), such as TOC and manganese. The longevity of GAC media, IX resin, and membrane operations are significantly affected by the quality of the source. Differences in source water quality parameters not specifically included (e.g., TOC, sulfate, pH, alkalinity, etc.) with pertinence to design or performance were reflected in cost by varying design parameters and treatment system performance according to probability functions using Monte Carlo analysis. This is primarily controlled through variation of the treatment performance factors (e.g., EBCT, surface area loading rate) to reflect less or more challenging water quality characteristics. The methodology for the Monte Carlo Simulation is covered in Section 5.2. Work is in progress to estimate costs associated with removing manganese and will be made available at a later date.

5.2 Monte Carlo Simulation for Design and Performance Variability

Water treatment system design is a practice that evolves non-uniformly across the country. Decisions in the design process are driven in some cases by rigorous engineering standards and in others by regional and geographic considerations, or owner and operator preferences. The result is a landscape of treatment systems across the United States that cannot be effectively modeled by clear and simple rules and frameworks. Additionally, water quality characteristics vary both regionally and locally, and these variations cannot be fully captured in the model with distinct data. These water quality characteristics

may improve or hinder performance as well as increase costs to ensure water quality downstream is not altered and complies with other regulations.

To compensate for this uncertainty, Monte Carlo methods were applied to simulate variation and to account for unknowns in major factors influencing design, operation, and, ultimately, cost for PFAS reduction systems. The @RISK Probabilistic Risk Analysis Software by Lumivero, which functions through an Excel add-in, was utilized for the Monte Carlo analysis.

Monte Carlo methods consist of randomizing inputs (e.g., loading rate, GAC media life, RO recovery) according to a defined distribution and number of iterations while calculating the impact to the outputs (e.g., number of vessels, media replacement frequency, cost). As the number of variables undergoing Monte Carlo analysis increases, computer processing power and the time to simulate one scenario both increase exponentially. Thus, Monte Carlo analysis was limited to only major factors considered to exert significant influence on design, performance, and cost of the individual systems. The major factors subjected to Monte Carlo are shown in Table 5-4.

Table 5-4 Major Factors for Monte Carlo Analysis

Parameter	Value
GAC - Pressure	
Surface Loading Rate	
Distribution Type	Triangular
Minimum Value	4 gpm/sf
Maximum Value	10 gpm/sf
Most Likely Value	6 gpm/sf
EBCT	
Distribution Type	Triangular
Minimum Value	10 min
Maximum Value	20 min
Most Likely Value	18 min
GAC - Basins	
Surface Loading Rate	
Distribution Type	Triangular
Minimum Value	4 gpm/sf
Maximum Value	10 gpm/sf
Most Likely Value	4 gpm/sf
EBCT	
Distribution Type	Triangular

Parameter	Value
Minimum Value	10 min
Maximum Value	20 min
Most Likely Value	18 min
GAC Bed Volumes to Breakthrough (Note 1)	
Distribution Type	Triangular
Minimum Value	75 percent of prediction
Maximum Value	175 percent of prediction
Most Likely Value	Prediction
IX - Vessels	
Surface Loading Rate	
Distribution Type	Triangular
Minimum Value	5 gpm/sf
Maximum Value	12 gpm/sf
Most Likely Value	8 gpm/sf
EBCT	
Distribution Type	Triangular
Minimum Value	1.5 min
Maximum Value	3 min
Most Likely Value	2 min
IX Bed Volumes to Breakthrough (Note 1)	
Distribution Type	Triangular
Minimum Value	75 percent of prediction
Maximum Value	175 percent of prediction
Most Likely Value	Prediction
Reverse Osmosis/Nanofiltration	
Surface Loading Rate	
Distribution Type	Uniform
Minimum Value	70 percent
Maximum Value	85 percent
Notes:	

Parameter	Value
1. GAC and IX Performance (i.e. determination of media life) is described in Section 5.4.1. Predicted value is determined using the generalized logistic function of the Clark model.	

With the exception of RO recovery, all Monte Carlo inputs were assigned a triangular distribution. A triangular distribution is a probability distribution where the probability decreases linearly on either side of the most likely value (highest probability) to the minimum and maximum, at which point the probability is zero. Triangular distributions were used where typical industry design values exist. RO recovery was modeled using a uniform distribution where each value between the minimum and maximum have an equivalent probability of occurrence.

The result of the Monte Carlo analysis is a distribution of possible costs for each technology (i.e., low [10th percentile], high [90th percentile], and most probable). For each modeled scenario, each of these costs was stored as a modeled output for each system represented in the occurrence database for use in determining the overall national cost of compliance with the modeled limit.

5.3 Capital Cost Calculation

Capital costs were calculated for each EPTDS of a PWS based on the design flow per EPTDS (refer to Equation 1). The design flow was used for capital costs estimates since equipment should be sized for peak treatment flow rates. Costs were independently calculated for IX, GAC vessels, GAC basins, and RO as described in the following subsections. Capital costs generated for individual systems represent a Class 5 Association for the Advancement of Cost Engineering (AACE) estimate, at approximately 1 to 2 percent maturity level of deliverable definition.

5.3.1 Major Hardware Components

5.3.1.1 GAC Gravity Basins

The major cost components incorporated into the capital cost estimate for this option are the concrete basins themselves, an influent pump station, media for the initial fill, and a building to house the system. The design assumptions for each element are summarized in Subsection 3.1.2.

The concrete basin includes costs for influent and effluent piping, isolation valves, and monitoring instruments. Using the design flow rate and the SLR, a required surface area for filtration is calculated and used to determine the appropriate number of basin cells and anticipated basin dimensions for costing.

Once number and size of basins are calculated, the design flow and specified EBCT is used to determine the volume of media needed. Cost of media was determined by converting volume to mass using an average GAC density of 0.5 g/cc and an average cost per pound of $1.40. It should be noted that cost changes were not projected into the cost model resulting from increased demand for adsorbent media.

The pump station includes costs for influent pumps, backwash pumps, an influent wetwell, and a backwash recovery basin. The independent design inputs for the influent pumps are total dynamic head (TDH) and total number of pumps. The independent design parameters for backwash pump and backwash recovery basin calculations are backwash loading rate, backwash duration, backwash frequency, and backwash pump TDH. Costs for backwash pumping include a single duty pump and a single standby pump.

The sum of the square footage required for the contactor basins was multiplied by a sizing factor of two to account for the ancillary equipment and space for access and maintenance. Pump station square footage, including all pumps and the wet well, was estimated by benchmarking design flow against previous designs. Building area was assumed to be the sum of contactor facility area (including sizing factor), pump station area, and backwash recovery basin area (assumed to be indoors). The building cost was assumed to be $200/sf.

Black & Veatch utilized empirically derived cost curves as a function of size from several decades of infrastructure project design and delivery to estimate cost for these major components. A curve for concrete basins provides cost as a function of square footage. A curve for steel tanks provides costs as a function of volume in gallons, and a curve for pumps provides cost as a function of horsepower.

Installation fees were included at 20 percent for all major equipment components, as summarized in Table 5-5. These cost factors are identical to those for GAC and IX pressure vessels.

Table 5-5 GAC and IX Equipment Installation Cost Factors

Component	Percent Multiplier of Unit Cost
Basins/Pressure Vessels	20%
Influent Pumps	20%
Backwash Pumps	20%
Influent Wetwell	20%
Backwash Recovery Basin	20%

5.3.1.2 GAC, IX and Manganese Pretreatment Pressure Vessels

Capital equipment costs were calculated using the total contactor footprint, contactor building footprint, and media volume required. Capital costs were calculated for the ancillary pump stations using the building footprint, number and size of influent pumps, backwash pumps, influent wetwell, and backwash recovery basin. The model incorporated a building cost of $200/ft^2. The installation fees for the various components are the same as those summarized in Table 5-6.

Calculated capital cost for manganese pretreatment for each system was considered a stand-alone output and was not included in the capital, operational, or life-cycle cost outputs for PFAS treatment.

5.3.1.3 Reverse Osmosis

Capital costs were calculated for the RO system and building, low- and high-pressure feed pumps and their associated building, storage tanks, cartridge filters, chemical treatment system, decarbonation system, and brine disposal. The model incorporated a building cost of $200/ft^2. The installation fees for the various components are summarized in Table 5-6.

Table 5-6 RO Equipment Installation Cost Factors

System	Component	Percent Multiplier of Unit Cost
Storage Tanks	RO Feed Tank	15%
	CIP Tank	15%
	CIP Neutralization Tank	15%
	Flush Tank	15%
Pump Stations	RO Low Pressure Feed Pumps	25%
	RO High Pressure Feed Pumps	25%
	CIP Pumps	20%
	Flush Pumps	20%
Cartridge Filter	RO Feed Cartridge Filter	20%
	CIP Cartridge Filter	20%
Chemical Feed Systems	Antiscalant	20%
	Sulfuric Acid	20%
	Caustic/Liquid Lime	20%
Decarbonation System	All related equipment	20%

5.3.1.4 Additional Capital Costs

In addition to equipment costs, the capital costs for GAC, IX, RO, and manganese pretreatment included additional project costs (site work, yard piping, electrical, and instrumentation and controls), contractor markup costs, and non-construction costs. The multipliers used for each of these factors are summarized in Table 5-7.

Table 5-7 Additional Capital Cost Assumptions

Additional Capital Costs	Description	Percent Multiplier of Total Equipment Costs
Additional Project Costs	Site Work	8.0%
	Yard Piping	9.0%
	Electrical	10.0%
	Instrumentation & Controls	2.5%

Additional Capital Costs	Description	Percent Multiplier of Total Equipment Costs
Contractor Markup Costs	Overhead	7.0%
	Profit	10.0%
	Mobilization/Bonds/Insurance	3.0%
	Contingency	4.0%
Non-Construction Costs	Permitting	1.0%
	Engineering	8.0%
	Legal/Administration	0.5%
	Construction Services	7.0%
	Commissioning/Startup	3.0%
	Contingency	30.0%

5.4 Operating Cost Calculation

The operational costs for GAC, IX, and RO were calculated using the average flow rate for each EPTDS, as represented by the average flow per water system divided by the number of EPTDS. Whereas capital costs were driven by maximum PFAS levels, the operating costs incurred were driven by the average influent PFAS concentrations to reflect long-term operating conditions. The tool allows entry of a treatment goal expressed as a percent of the potential regulatory limit, and the resulting target concentration serves as the effluent concentration trigger for replacement of media. This target may be expressed either as a concentration of a single PFAS compound or as a combination of compounds.

Operating costs that were considered for this work included replacement costs (using the calculated bed volumes to breakthrough or media replacement frequency), power consumption in the pumps and buildings, maintenance costs, waste disposal, and labor costs. Analytical monitoring costs were not included in the life-cycle cost calculations. Table 5-8 provides an overview of the O&M cost assumptions.

Table 5-8 O&M Cost Assumptions

O&M Category	Description	Value
Media Replacement	GAC Virgin Media(Note 1)	$1.40/lb
	GAC Reactivated Media	$1.20/lb
	IX Resin	$240/ft^3 ($3.70/lb @ apparent density of 1.05 g/cc)
Membrane Replacement	Membrane Cost	$600/element

O&M Category	Description	Value
Power	Unit Cost	$0.10/kilowatt-hour (kWh)
	Unit Building Power Usage	19.5 kWh/ft^2/yr
	Building Utilization Factor	365 days/year
Maintenance	Installed Equipment	1.5% Percent Multiplier of Capital Costs
	Structures and Facilities	1.0% Percent Multiplier of Capital Costs
Waste Disposal	Incineration(Note 2)	$720/ton
	GAC Density	0.5 g/cc
	IX Density	1.05 g/cc
	Mn Adsorptive Media Density	1.8 g/cc
Chemical Consumption Costs	Antiscalant	$15.00/gal
	Sulfuric Acid	$2.50/gal
	Caustic	$4.50/gal
Labor	Operator Rate	$30/hr
	Admin Rate	$25/hr
	Number of Valves	3 per vessel/basin, 2 per pump (additional requirements for RO system include 2 per cartridge filter, 3 per decarbonation system, and 2 per tank)
	Number of Instruments	2 per vessel/basin, 1 per pump (additional requirements for RO system include 2 per cartridge filter, 2 per decarbonation system, and 1 per tank)
	Record Keeping and Sampling	5 minutes per day per instrument
	Pump Operation (adjustments)	5 minutes per day per pump
	Valve Adjustments	5 minutes per week per valve
	GAC Contactor Maintenance	1 hour per week per vessel/basin
	IX Replacement	16 hours per bed volume
	Cartridge Filters	12 hours per year per cartridge filter
	RO Membrane Process Labor	120 hours per week

Notes:

1. Life-cycle cost factors were chosen to match the EPA's standard practice for estimating life-cycle cost
2. Spent GAC media and IX resin was assumed to be incinerated because of the unknown viability of GAC media reactivation under CERCLA. Replacement costs were therefore assumed to be virgin media.

5.4.1 Estimation of Media Life and Disposal

The generalized logistic function of the Clark model (Clark, 1987), represented in Equation 2, was the basis for calculations for estimation of media life for both GAC and IX. While more rigorous techniques exist for modeling adsorption, Clark's model was utilized for its relative simplicity and accuracy.

$$C = \frac{{C_o}^{n-1}}{1 + Be^{-r\prime t}} \qquad (2)$$

Where:

C_o is the influent contaminant concentration, C is the concentration of a given contaminant at time t, n is the inverse of the slope of the Freundlich isotherm, and r' and B are constants. Rearranging the equation above to:

$$\ln\left[\left(\frac{C_o}{C}\right)^{n-1} - 1\right] = -r't + ln\ B$$

r' and B can be solved for from the slope and intercept of the plot of $\ln[(C_o/C)^{1/n}\text{-}1]$ versus time. If a constant flow is assumed, the number of bed volumes becomes directly proportional to time, allowing these relationships to be expressed as a function of bed volumes treated rather than time. B, n, and r' values utilized for GAC and IX are expressed in Table 5-9. The values utilized for GAC were derived from data collected during a Black & Veatch GAC pilot study for 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 (LHHCWD).

Table 5-9 Values Variables in Modeled Bed Life

Media	Constant	PFOA	PFOS	PFHxS	PFNA	PFHpA	PFBS
IX(Note 1)	*n*	1.25	1.25	1.25	1.25	1.25	1.25
	B	4.8	1.8	1.4	6.0	3.6	3.2
	-r'	-2.55E-05	-3.33E-06	-3.40E-06	-1.70E-05	-2.62E-05	-6.23E-06
GAC	*n*	1.49	1.54	3.23	1.79	1.67	1.56
	B	141.7	15.8	666.0	49.1	49.1	11.3
	-r'	-6.21E-04	-2.07E-04	-3.77E-04	-3.46E-04	-4.81E-04	-3.28E-04

Notes:

1. Parameters for PFOA and PFHpA were derived from the CFPUA data set. Parameters for PFBS, PFHxS, and PFOS were derived from the LHHCWD data set. Parameters for PFNA were estimated by extrapolating data for PFOA and PFHpA because insufficient pilot data were available to support a curve fit determination.

For each system with occurrence data, *C* was calculated for each PFAS compound at a specified bed volume increment. Increments of 250 bed volumes up to a maximum of 40,000 were calculated for GAC. Increments of 5,000 bed volumes up to a maximum of 800,000 were calculated for IX. The number of bed volumes at which *C* exceeded the specified target replacement concentration was determined, and the number of bed volumes for the first contaminant to breach its target concentration was used to calculate media replacement frequency. The number of bed volumes treated before the first contaminant exceeded the target concentration was subjected to Monte Carlo variability as described in Section 5.2.

5.5 Life-Cycle Costs

The model determines 20-year life-cycle costs, which combines the capital costs and annual operating and maintenance costs. Life-cycle costs provide a means of comparing the costs of alternative technologies over the life cycle of the equipment. The life-cycle costs in the body of this report were calculated assuming a 20-year lifespan and a discount rate of 3 percent. A comparison of annualized NPDWR costs by system size at 3 and 7 percent is included in Table A-5 of Appendix A . While typical practice to determine life-cycle costs may incorporate other factors, such as the inflation and loan interest, the discount rate was used to match the approach that is standard practice for the EPA in promulgating national primary drinking water regulations.

6.0 National Cost Assessment Methodology

The conceptual framework for assessing the national costs is as follows:

- Assess capital, annual O&M, and life-cycle costs for each EPTDS in every water system for which potential regulatory limits for PFAS may require treatment.
- Average the costs by system size category and system type (ground or surface water).
- Multiply those average costs by the total anticipated number of systems of each type impacted in each system size category based on the percentage of systems in the database impacted by a proposed regulatory limit for PFAS.

The following subsections summarize the process and details associated with the national cost estimation methodology.

6.1 Estimating National Costs Using Model Outputs

Due to the difference in number of EPTDS for groundwater and surface water systems, the national cost calculations were completed separately for groundwater and surface water systems. The following methodology was utilized for each source water classification:

1. Using the treatment facility costs for systems from the occurrence database, the costs were binned by system size, and average EPTDS costs per system size bin were calculated.

2. Using the occurrence database, the number of impacted systems per size category was calculated, and the corresponding percent of the systems in the database was determined.

3. To estimate the number of impacted systems nationally, the percentage of impacted systems in the occurrence database was multiplied by the total number of systems in SDWIS for each size category.

4. The estimated number of impacted systems per size category multiplied by the average cost per EPTDS and the assumed number of entry points yields the total cost per size category. The sum of all costs per size category yields the estimated national cost of removing PFAS to a potential regulatory limit.

A summary output is included in Table 6-1, which displays the costs associated with a achieving a maximum contaminant level (MCL) in drinking water of 4 ppt for PFOA and 4 ppt for PFOS for groundwater systems only.

Table 6-1 Example Summary Capital Cost Table for an MCL of 4 ppt for PFOA and PFOS (Groundwater Systems Only)

Size Category	PWSs In Database (Note 1)	Impacted PWSs in Database	% Impacted in Database	Active PWSs in SDWIS (Note 1)	Estimated Nationally Impacted PWSs	EPTDS per PWS	Average Capital Cost per EPTDS	National Capital Cost
1	1262	264	21%	10,654	2,229	1	$900,000	$2,006,100,000
2	1031	191	19%	13,037	2,415	1	$1,900,000	$4,588,500,000
3	285	37	13%	4,132	536	2	$2,500,000	$2,680,000,000
4	301	40	13%	5,503	731	2	$3,200,000	$4,678,400,000
5	373	65	17%	2,784	485	2	$5,300,000	$5,141,000,000
6	1221	106	9%	1,385	120	4	$8,500,000	$4,080,000,000
7	171	19	11%	162	18	10	$12,500,000	$2,250,000,000
8	78	11	14%	74	10	12	$17,700,000	$2,124,000,000
9	2	2	100%	2	2	39	$40,600,000	$3,166,800,000
All Systems	4724	735	16%	37,733	6,546	N/A	N/A	$30,714,800,000

Notes:

1. The number of PWSs was updated per information included in EPA's March 2023 proposed regulation package and accounts for only community water systems (CWSs), which are PWSs that serve more than 25 people year round.

6.2 Accounting for State Level Regulatory Costs

The model includes consideration of state regulatory actions that may have driven PWSs to remove PFAS already. Consideration of state regulatory actions is necessary to characterize the compliance costs of a potential NPDWR for PFAS. All state regulations incorporated into modeled cost output are shown in Table 6-2

Table 6-2 State Maximum Contaminant Levels Modeled for State Regulatory Cost Estimate

States	Type	PFOA	PFOS	PFHxS	PFNA	PFHpA	PFBS
Connecticut	Individual		10				
Delaware	Individual	14	21				
Delaware	Combined	17	17				
Massachusetts	Combined	20	20	20	20	20	
Michigan	Individual	8	16	51	6		
New Hampshire	Individual	12	15	18	11		
New Jersey	Individual	14	13		13		
New York	Individual	10	10				
Ohio	Combined	70	70				
Ohio	Individual			140	21		140,000
Vermont	Individual	20	20	20	20	20	
Wisconsin	Combined	70	70				

To differentiate federal regulatory costs from costs incurred because of existing state regulations, the cost tool includes an input sheet for all existing state MCLs as either individual limits or group totals. The Visual Basic Script references both the state MCLs and the projected federal MCLs. In the absence of a federal regulation (or if the state MCL is more stringent than the federal MCL), the cost tool generates costs for treatment to comply with the state MCLs on the input sheet. An example of this is shown in Table 6-3, which displays treatment costs incurred as a result of state regulations shown in Table 6-2.

Table 6-3 Summary of Estimated Costs Associated with State PFAS MCLs

PWS Size Category	Population Range	% Impacted	Estimated Number of Impacted PWSs	Average CAPEX/PWS	Average O&M/PWS	Annualized PWS Cost	Annualized Total Cost	Present Value of Lifecycle Cost[1]
1	25 to 100	6%	718	$800,000	$20,000	$74,000	$53,132,000	$790,500,000
2	101 to 500	5%	763	$1,700,000	$30,000	$145,000	$110,635,000	$1,646,000,000
3	501 to 1,100	4%	232	$4,400,000	$68,000	$364,000	$84,448,000	$1,256,400,000
4	1,001 to 3,300	1%	111	$5,300,000	$100,000	$457,000	$50,727,000	$754,700,000
5	3,301 to 10,000	5%	240	$8,600,000	$200,000	$779,000	$186,960,000	$2,781,500,000
6	10,001 to 50,000	3%	99	$20,400,000	$300,000	$1,672,000	$165,528,000	$2,462,600,000
7	50,001 to 100,000	1%	6	$46,200,000	$800,000	$3,906,000	$23,436,000	$348,700,000
8	100,001 to 1,000,000	3%	11	$50,200,000	$1,100,000	$4,475,000	$49,225,000	$732,300,000
9	>1,000,000	4%	1	$1,095,900,000	$23,010,000	$96,672,000	$96,672,000	$1,438,200,000
All Systems		4%	2181	N/A	N/A	N/A	$820,763,000	$12,210,900,000

Notes:

1. Present value of lifecycle costs estimated based on 3% discount rate and 20-year lifespan of equipment.
2. National costs for two potential MCLs are summarized in Section 7.1. The differentials between state costs in this table the total national costs represent the cost associated with any modeled NPDWR.

7.0 Summary of Results

A summary of the cost model results for various potential federal MCL alternatives on the national and household level is presented in this section.

7.1 National Cost Estimates

The national cost modeling tool was used to evaluate both the national financial burdens on communities from PFAS drinking water contamination (the National Burden) and the costs for water systems to comply with a potential NPDWR for PFAS (NPDWR Compliance Costs).

The National Burden is reflective of the total, cumulative impact to water systems and communities across the United States from PFAS contamination of drinking water. It is calculated by estimating the drinking water PFAS treatment costs associated with the number of systems with PFAS occurrence data above the target limit. The National Burden assumes the same target limit for water systems across all states and includes systems in states with existing drinking water regulations for PFAS. The NPDWR Compliance Costs are determined by estimating the national financial burden and excluding costs for systems already triggered into treatment by existing drinking water regulations at the state level. The difference between the National Burden and the NPDWR Compliance Costs is therefore calculated using the data presented in Table 6-3.

The National Burden and NPDWR Compliance Costs were estimated for two different scenarios. The first scenario is based on a target PFOA and PFOS level of 4 ppt each. The second scenario is based on target PFOA and PFOS levels of 10 ppt each.

An overview of the present value of the life-cycle cost for the National Burden and NPDWR compliance cost for each of these scenarios is displayed on Figure 7-1.

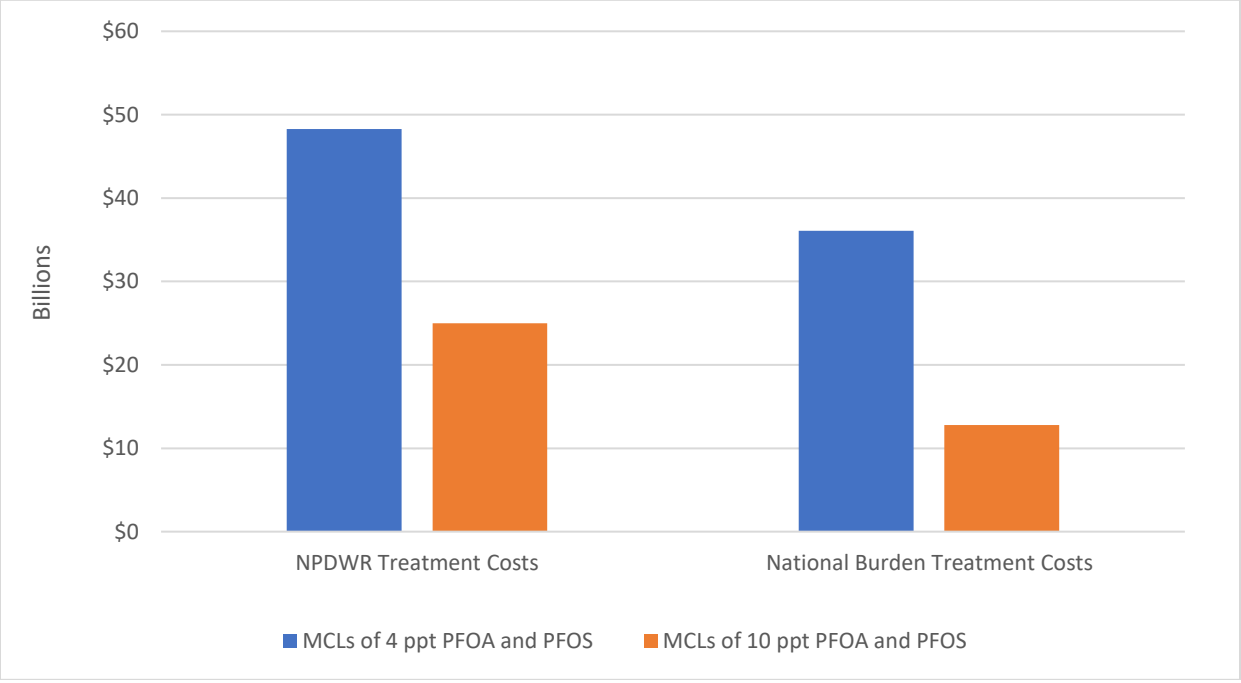


Figure 7-1 Summary of Present Value of Life-Cycle Costs for National Burdens and NPDWR Compliance Costs for Each Scenario based on a 3% discount rate

Annualized costs were also calculated using Formula 3. An overview of the National Burden and NPDWR Compliance Annualized Cost for each of these scenarios is presented on Figure 7-2.

$$Annualized\ Costs = \frac{(Capital\ Costs)(Discount\ Rate)}{1-(1+Discount\ Rate)^{-n}} + Annual\ Operating\ Costs \quad (3)$$

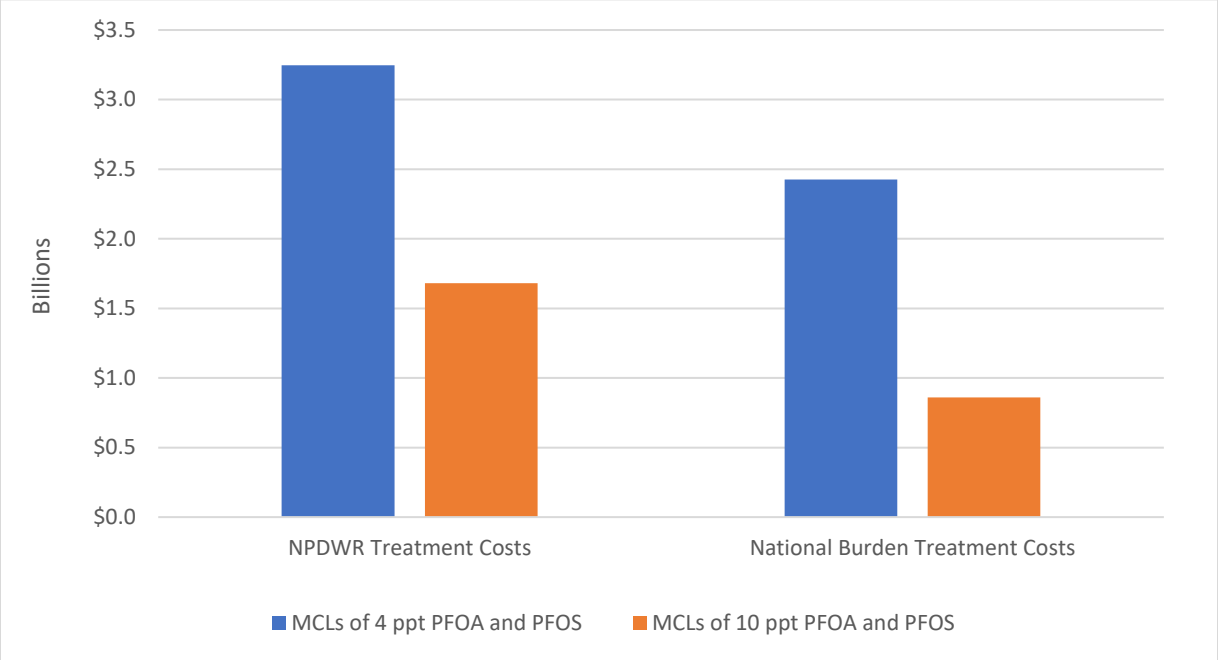


Figure 7-2 Summary of Annualized Costs for National Burdens and NPDWR Compliance for Each Scenario based on a 3% discount rate

A more detailed breakdown of these costs by system size is presented in Appendix A.

7.2 Household Financial Impacts

As part of this analysis, the annual financial impacts to individual households from costs associated with the installation and operation of drinking water treatment facilities for PFAS were determined. The financial impacts to individual households will vary by specific PFAS levels, system size, and other factors. Additionally, the impacts to individual households arising from a potential NPDWR will differ depending on whether there is an existing state regulation for PFAS in drinking water. Table 7-1 shows the individual household impacts as a function of system size for each of the three scenarios discussed in Section 7.1. These household level cost impacts are based on the annualized costs for each system size and an average of 2.6 persons per household and incorporate estimated average service populations for each size category based on SDWIS data. The range of household level costs in the table is reflective of communities where new treatment facilities will need to be installed and operated.

Table 7-1 Annual Costs to Household for Removing PFAS from Drinking Water

PWS Size Category	Population Range	Average Service Population	Approximate Range of Costs per Household
1	25 to 100	59	$3570 - $3570
2	101-500	245	$1675 - $1750
3	501-1,100	736	$1360 - $1390
4	1,001-3,300	1,939	$575 - $640
5	3,301-10,000	5,696	$305 - $325
6	10,001-50,000	20,613	$200 - $225
7	50,001-100,000	67,417	$155 - $175
8	100,001-1,000,000	204, 194	$65 - $70
9	>1,000,000	1,700,000	$115 - $120

Appendix A. Modeled Cost MCL and Discount Rate Comparison Tables

Table A-1 National Cost Burden by System Size for 4 ppt PFOA, PFOS (Groundwater Systems)

Size Category	Population Range	Estimated Number of PWSs Impacted	Estimated Number of EPTDS Impacted	Average CAPEX per EPTDS	Average OPEX per EPTDS	Total CAPEX	Total OPEX
1	25 to 100	2,229	2,229	$900,000	$20,000	$2,006,100,000	$44,580,000
2	101 to 500	2,415	2,415	$1,900,000	$30,000	$4,588,500,000	$72,450,000
3	501 to 1,100	536	1,072	$2,500,000	$39,000	$2,680,000,000	$41,808,000
4	1,001 to 3,300	731	1,462	$3,200,000	$41,000	$4,678,400,000	$59,942,000
5	3,301 to 10,000	485	970	$5,300,000	$82,000	$5,141,000,000	$79,540,000
6	10,001 to 50,000	120	480	$8,500,000	$156,000	$4,080,000,000	$74,880,000
7	50,001 to 100,000	18	180	$12,500,000	$215,000	$2,250,000,000	$38,700,000
8	100,001 to 1,000,000	10	120	$17,700,000	$388,000	$2,124,000,000	$46,560,000
9	>1,000,000	2	78	$40,600,000	$845,000	$3,166,800,000	$65,910,000
All Systems		6,546	9,006	N/A	N/A	$30,714,800,000	$524,370,000

Table A-2 National Burden Costs per System Size for 4 ppt PFOA, PFOS (Surface Water Systems)

Size Category	Population Range	Estimated Number of PWSs Impacted	Estimated Number of EDTDS Impacted	Average CAPEX per EPTDS	Average OPEX per EPTDS	Total CAPEX	Total OPEX
1	25 to 100	41	41	$900,000	$20,000	$36,900,000	$820,000
2	101 to 500	125	125	$1,900,000	$30,000	$237,500,000	$3,750,000
3	501 to 1,100	63	63	$2,600,000	$43,000	$163,800,000	$2,709,000
4	1,001 to 3,300	137	137	$3,400,000	$51,000	$465,800,000	$6,987,000
5	3,301 to 10,000	294	294	$5,500,000	$82,000	$1,617,000,000	$24,108,000
6	10,001 to 50,000	134	134	$9,200,000	$171,000	$1,232,800,000	$22,914,000
7	50,001 to 100,000	46	92	$13,000,000	$243,000	$1,196,000,000	$22,356,000
8	100,001 to 1,000,000	61	122	$18,600,000	$469,000	$2,269,200,000	$57,218,000
9	>1,000,000	2	8	$32,000,000	$1,270,000	$256,000,000	$10,160,000
All Systems		903	1,016	N/A	N/A	$7,475,000,000	$151,022,000

Table A-3 National Burden Costs by System Size for 10 ppt PFOA, PFOS (Groundwater Systems)

Size Category	Population Range	Estimated Number of PWSs Impacted	Estimated Number of EPTDS Impacted	Average CAPEX per EPTDS	Average OPEX per EPTDS	Total CAPEX	Total OPEX
1	25 to 100	861	861	$900,000	$20,000	$774,900,000	$17,220,000
2	101 to 500	835	835	$2,000,000	$30,000	$1,670,000,000	$25,050,000
3	501 to 1,100	290	580	$2,700,000	$33,000	$1,566,000,000	$9,570,000
4	1,001 to 3,300	201	402	$3,100,000	$37,000	$1,246,200,000	$7,437,000
5	3,301 to 10,000	246	492	$5,500,000	$71,000	$2,706,000,000	$17,466,000
6	10,001 to 50,000	84	336	$8,900,000	$119,000	$2,990,400,000	$9,996,000
7	50,001 to 100,000	12	120	$12,700,000	$170,000	$1,524,000,000	$2,040,000
8	100,001 to 1,000,000	10	120	$18,300,000	$277,000	$2,196,000,000	$2,770,000
9	>1,000,000	2	78	$42,100,000	$790,000	$3,283,800,000	$1,580,000
All Systems		2,541	3,824	N/A	N/A	$17,957,300,000	$93,129,000

Table A-4 National Burden Costs by System Size for 10 ppt PFOA, PFOS (Surface Water Systems)

Size Category	Population Range	Estimated Number of PWSs Impacted	Estimated Number of EPTDS Impacted	Average CAPEX per EPTDS	Average OPEX per EPTDS	Total CAPEX	Total OPEX
1	25 to 100	21	21	$900,000	$20,000	$18,900,000	$420,000
2	101 to 500	83	83	$2,000,000	$30,000	$166,000,000	$2,490,000
3	501 to 1,100	21	21	$2,800,000	$40,000	$58,800,000	$840,000
4	1,001 to 3,300	39	39	$3,700,000	$55,000	$144,300,000	$2,145,000
5	3,301 to 10,000	155	155	$5,600,000	$70,000	$868,000,000	$10,850,000
6	10,001 to 50,000	93	93	$9,400,000	$138,000	$874,200,000	$12,834,000
7	50,001 to 100,000	29	58	$13,400,000	$199,000	$777,200,000	$5,771,000
8	100,001 to 1,000,000	45	90	$19,800,000	$382,000	$1,782,000,000	$17,190,000
9	>1,000,000	1	4	$37,200,000	$1,100,000	$148,800,000	$1,100,000
All Systems		487	564	N/A	N/A	$4,838,200,000	$53,640,000

Table A-5 Annualized NPDWR Costs by System Size per Discount Rate

PWS Size Category	Population Range	4 ppt PFOA/PFOS		10 ppt PFOA/PFOS	
		3%	7%	3%	7%
1	25 to 100	$130,738,000	$169,422,000	$18,310,000	$23,682,000
2	101-500	$290,685,000	$387,667,000	$40,835,000	$55,309,000
3	501-1,100	$151,558,000	$200,989,000	$35,287,000	$51,609,000
4	1,001-3,300	$365,882,000	$489,646,000	$55,914,000	$77,610,000
5	3,301-10,000	$391,743,000	$518,888,000	$101,870,000	$142,992,000
6	10,001-50,000	$291,366,000	$381,145,000	$119,121,000	$169,335,000
7	50,001-100,000	$269,744,000	$355,852,000	$139,532,000	$194,533,000
8	100,001-1,000,000	$350,471,000	$454,900,000	$238,755,000	$331,892,000
9	>1,000,000	$209,464,000	$272,704,000	$136,734,000	$200,238,000
All Systems		$2,451,651,000	$3,231,213,000	$886,358,000	$1,247,200,000

Appendix C

Supplemental Figures Comparing Case Study Data with EPA and BV Cost Models

List of Contents

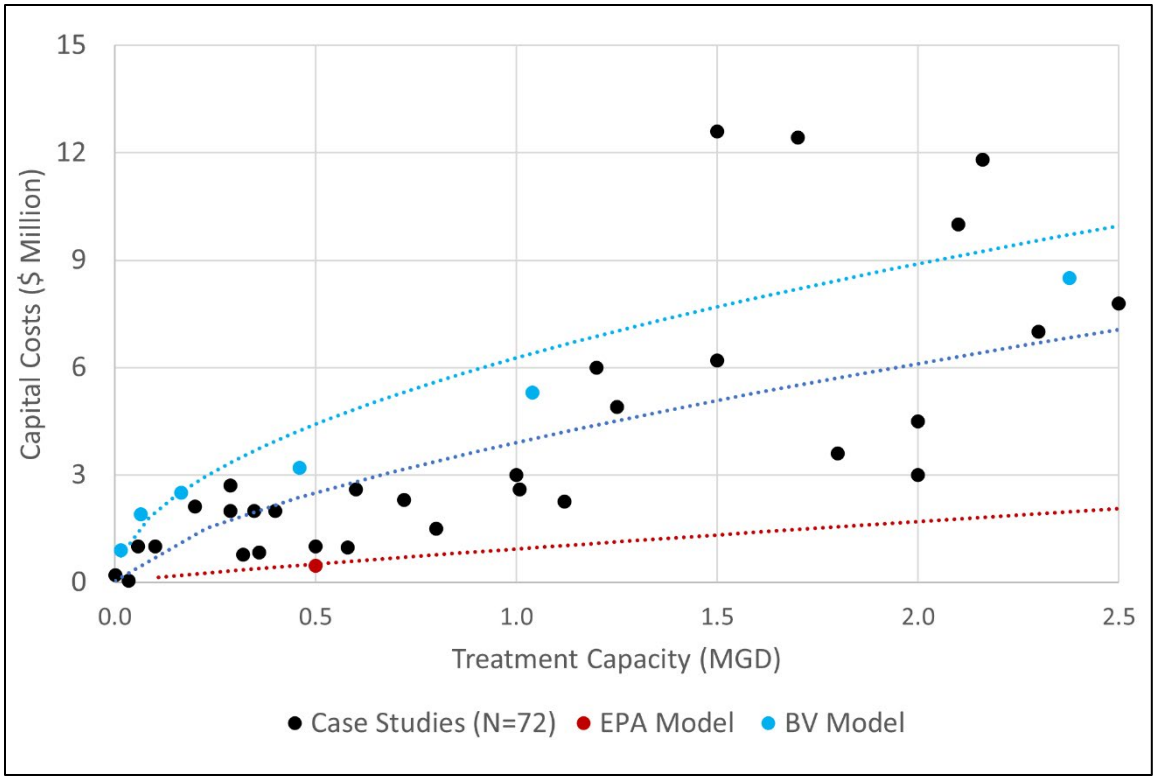


Figure A 13: Comparison of GAC Capital Costs for Smaller Systems (<2.5 MGD)

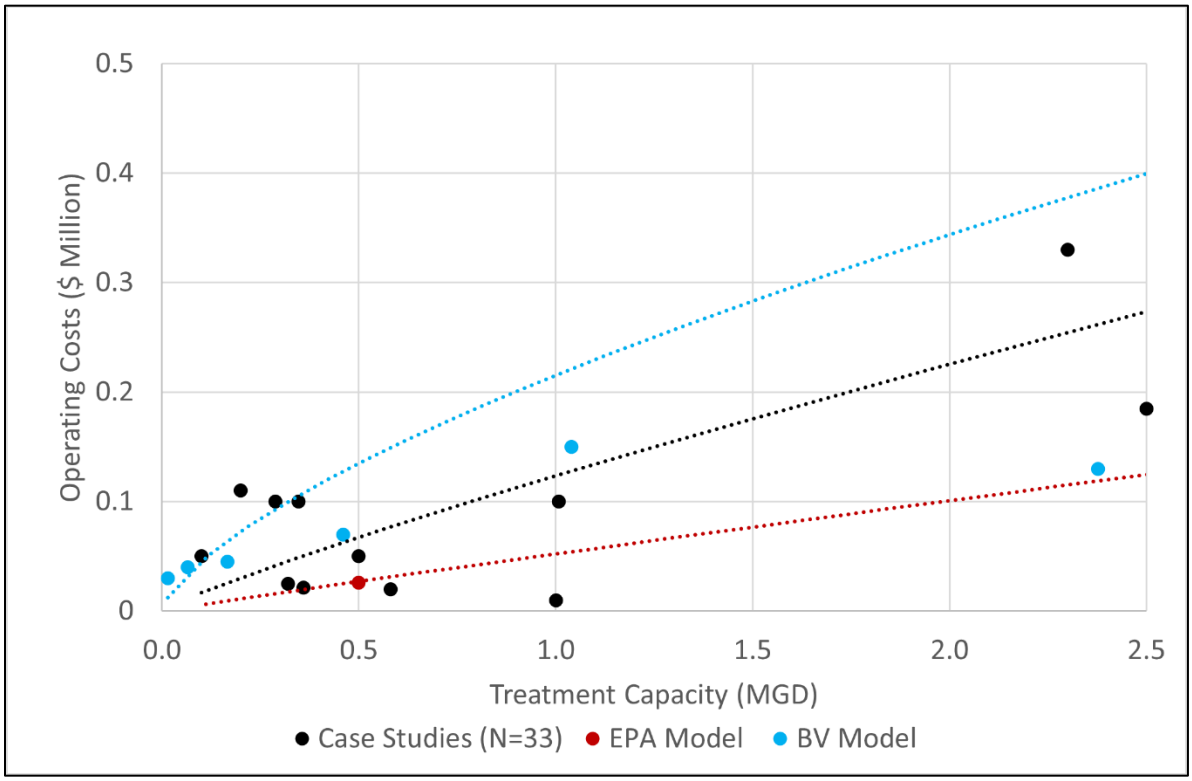


Figure A 14: Comparison of GAC Operating Costs for Small Systems (<2.5 MGD)

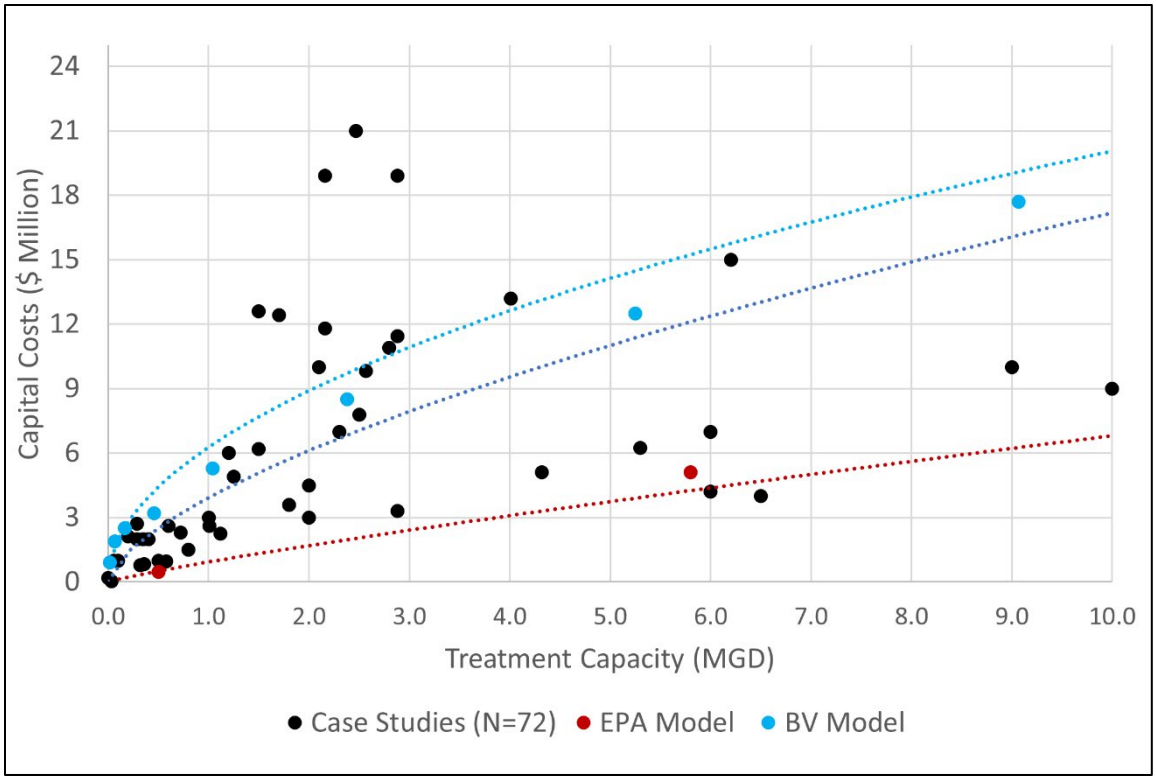


Figure A 15: Comparison of GAC Capital Costs for Medium Systems (<25 MGD)

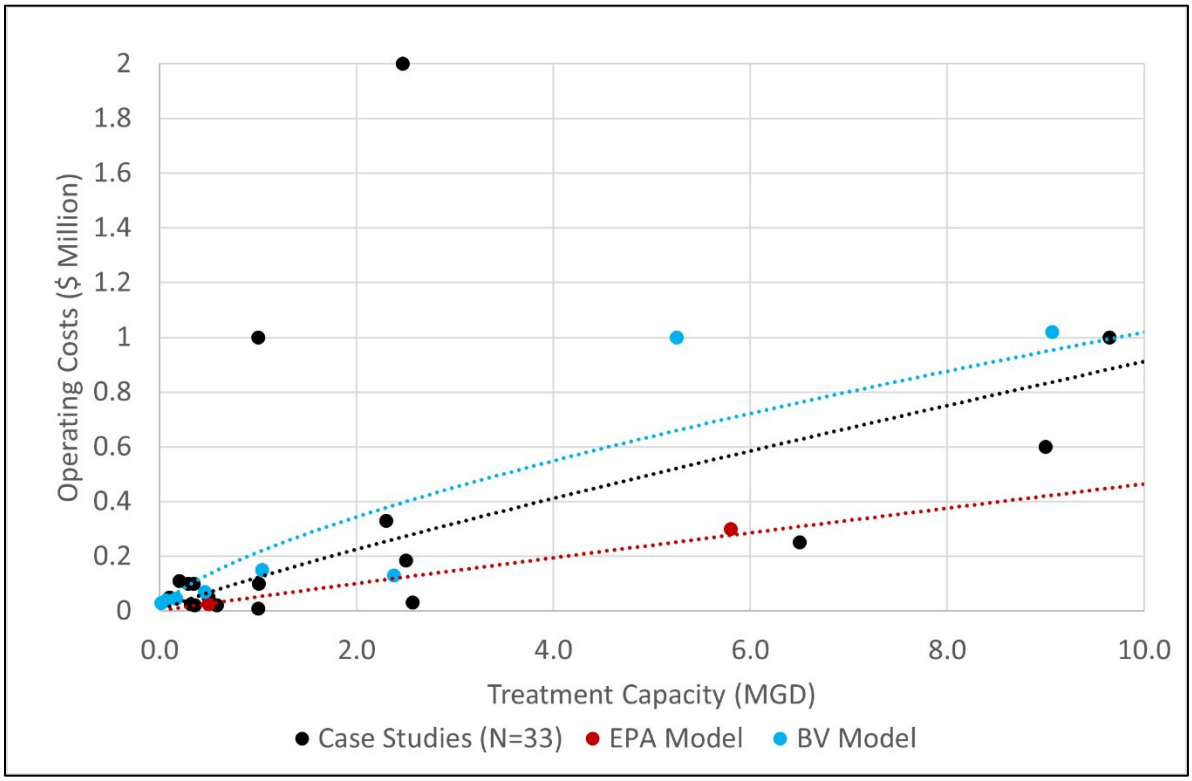


Figure A 16: Comparison of GAC Operating Costs for Medium Systems (<10 MGD)

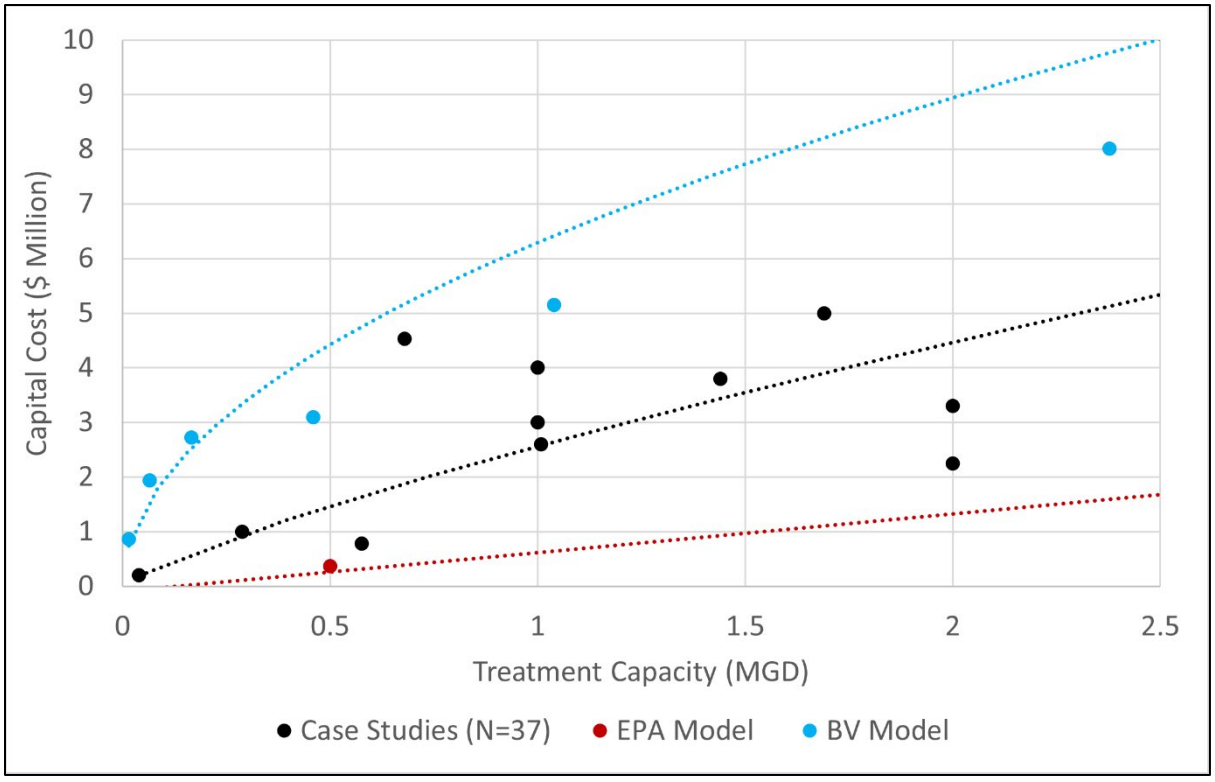


Figure A 17: Comparison of IX Capital Costs for Small Systems (<2.5 MGD)

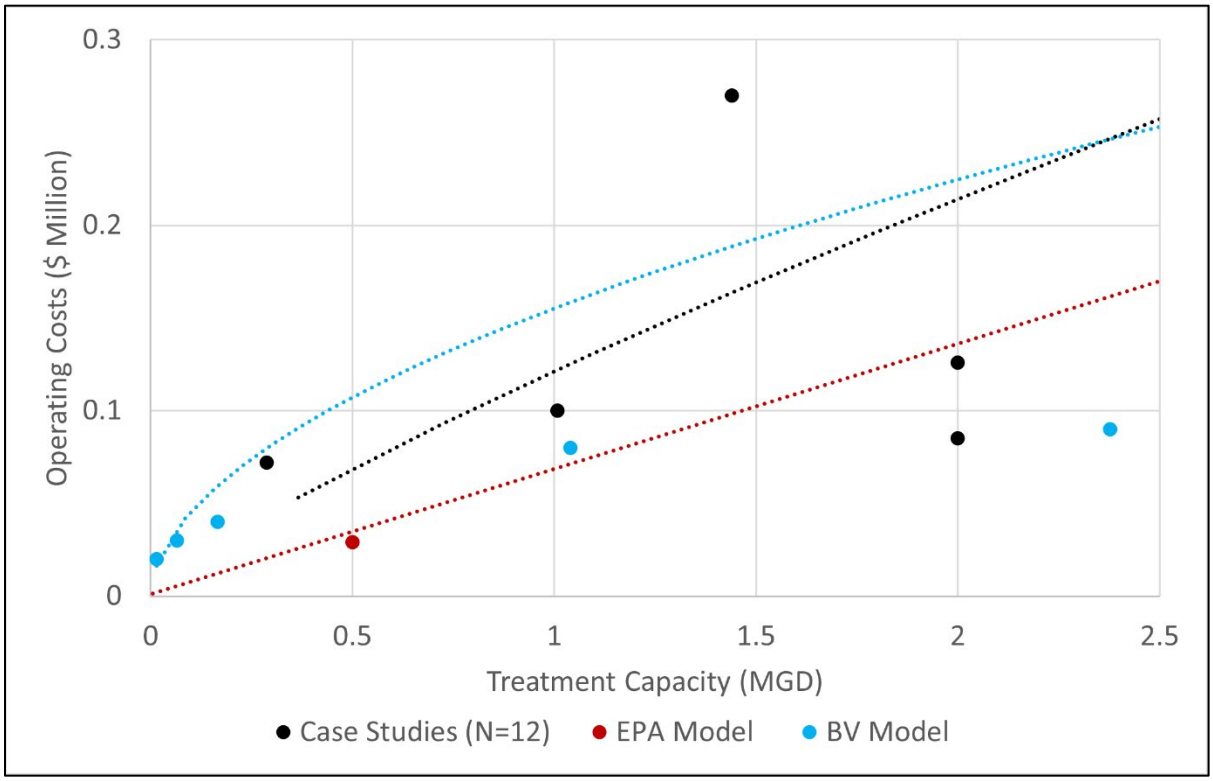


Figure A 18: Comparison of IX Operating Costs for Small Systems (<2.5 MGD)

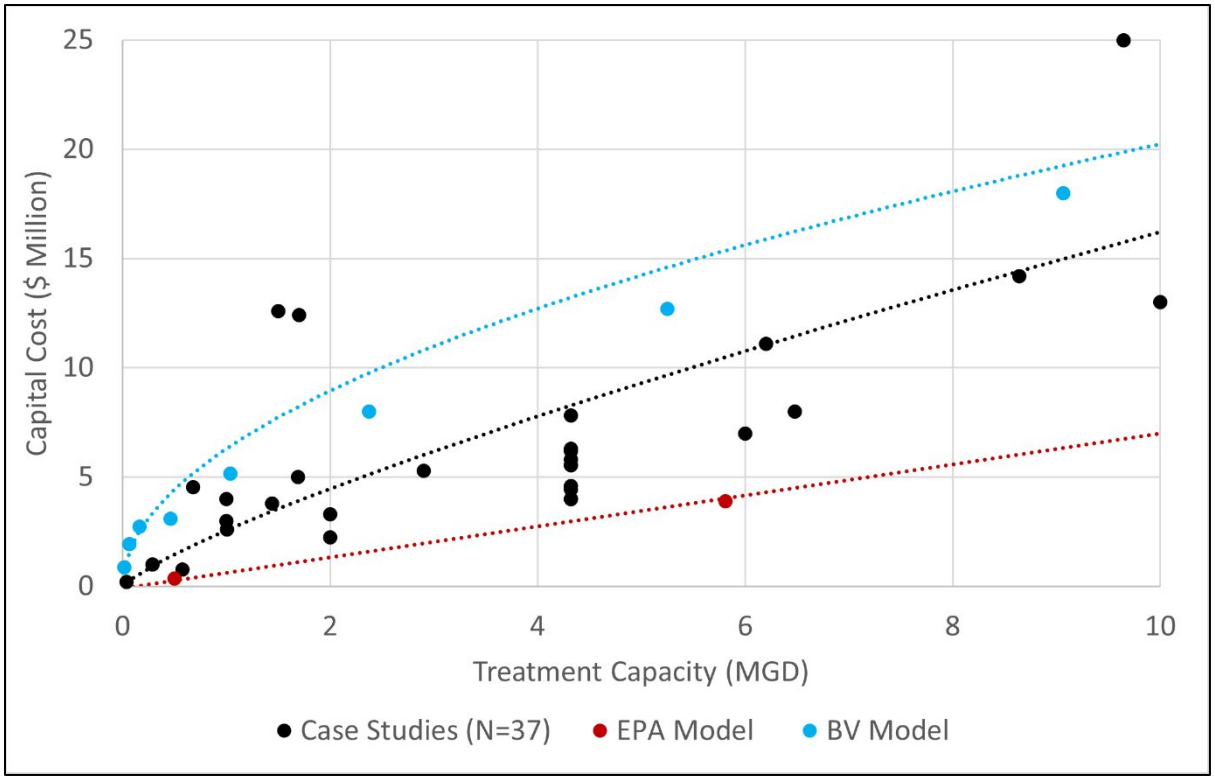


Figure A 19: Comparison of IX Capital Costs for Medium Systems (<10 MGD)

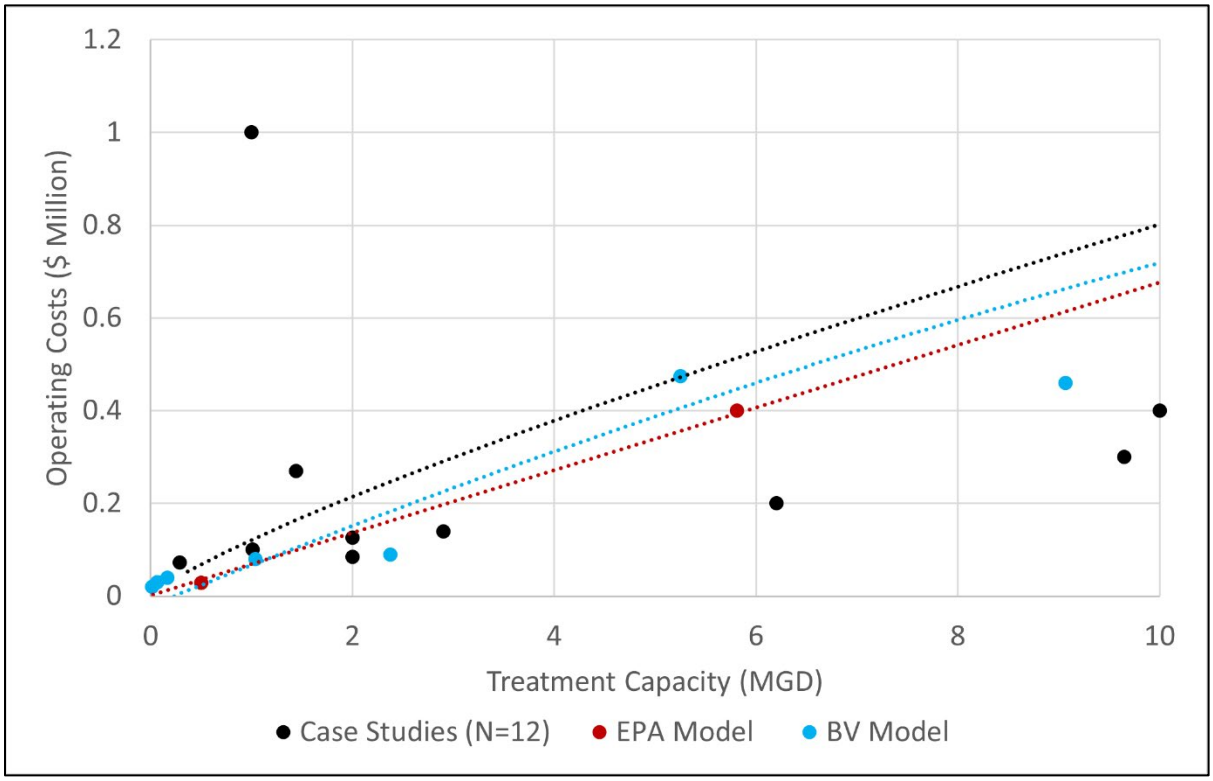


Figure A 20: Comparison of IX Operating Costs for Medium Systems (<10 MGD)

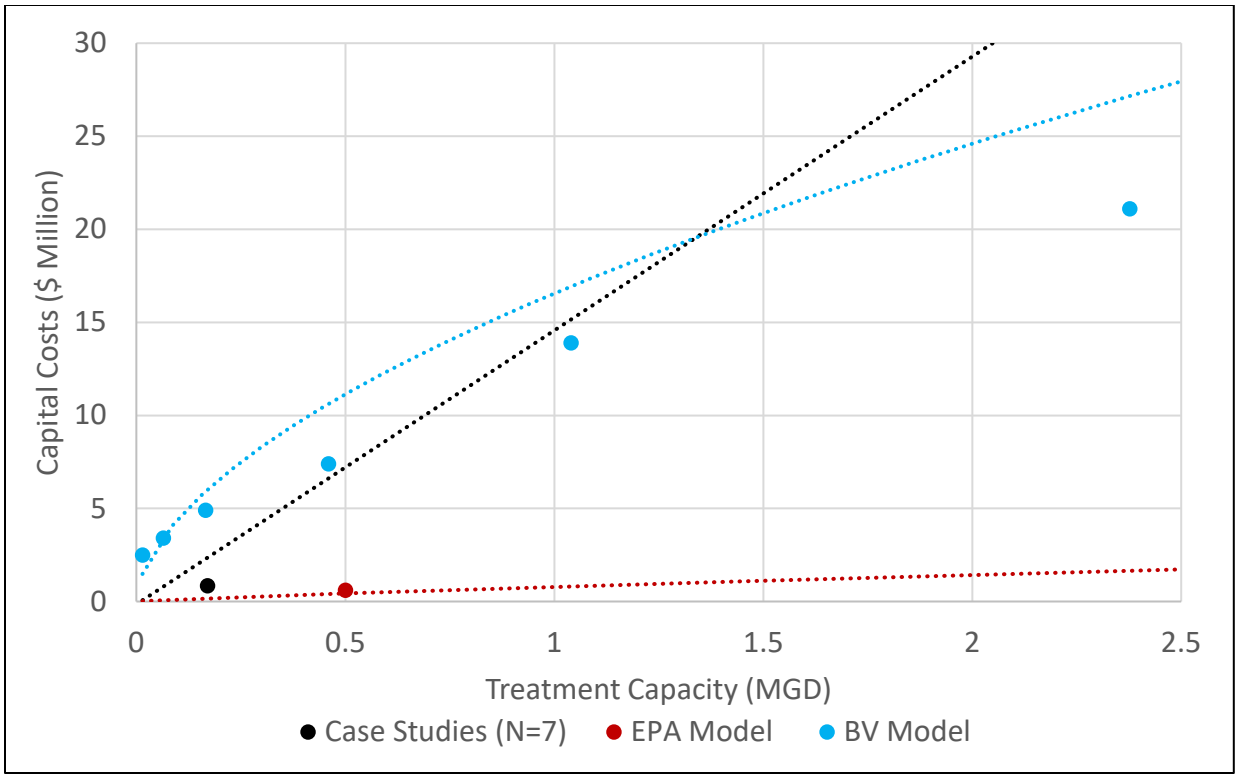


Figure A 21: Comparison of RO Capital Costs for Small Systems (<2.5 MGD)

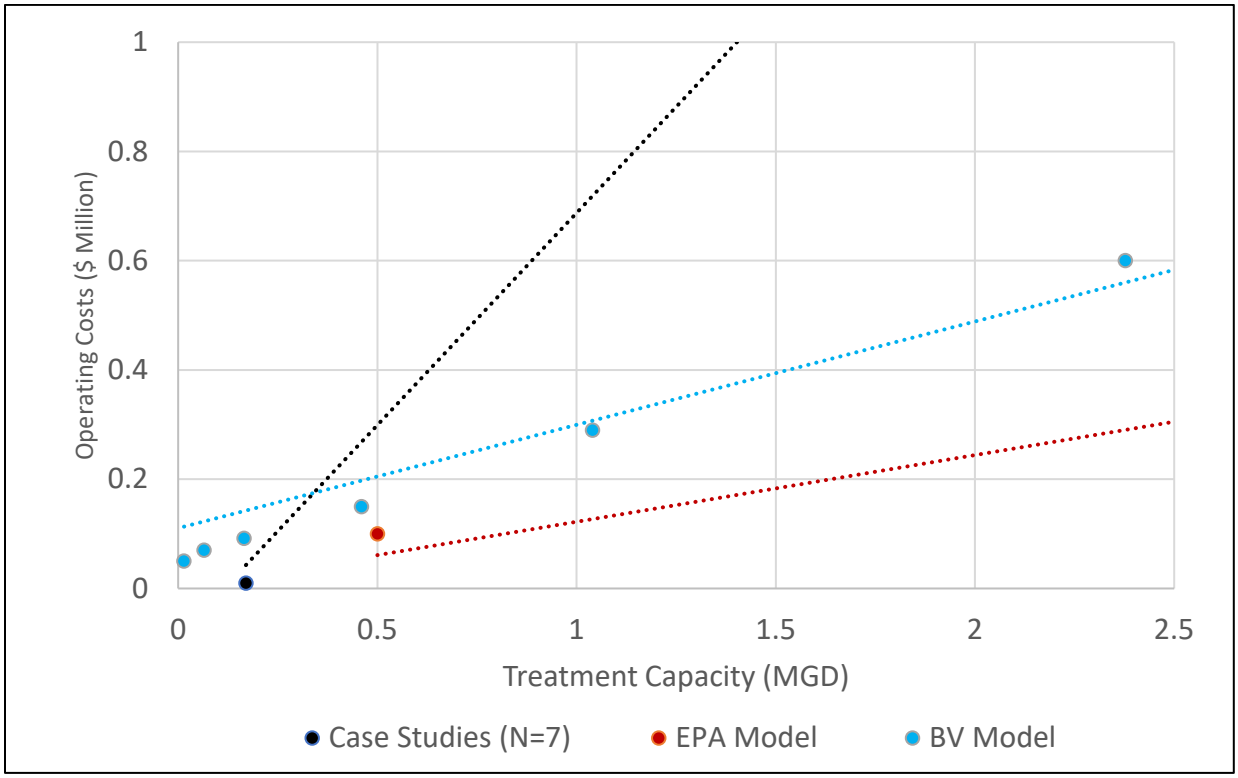


Figure A 22: Comparison of RO Operating Costs for Small Systems (< 2.5 MGD)

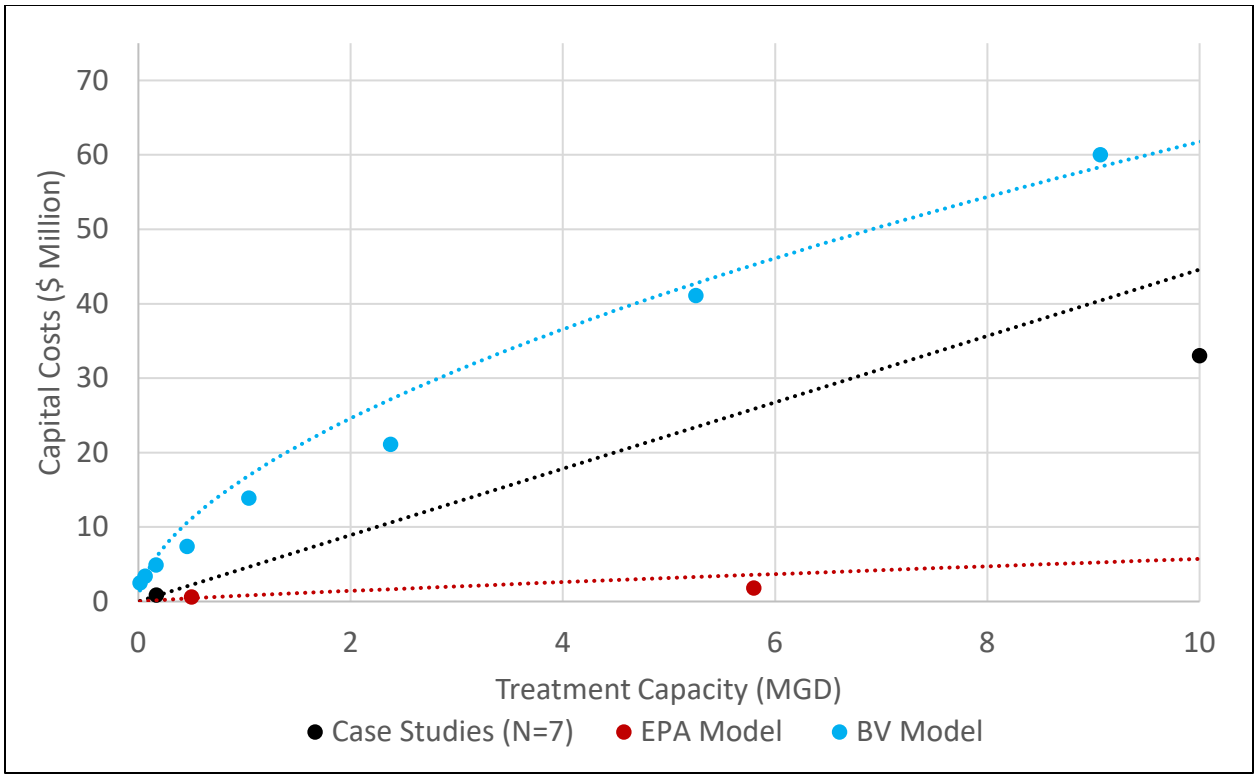


Figure A 23: Comparison of RO Capital Costs for Medium Systems (<10 MGD)

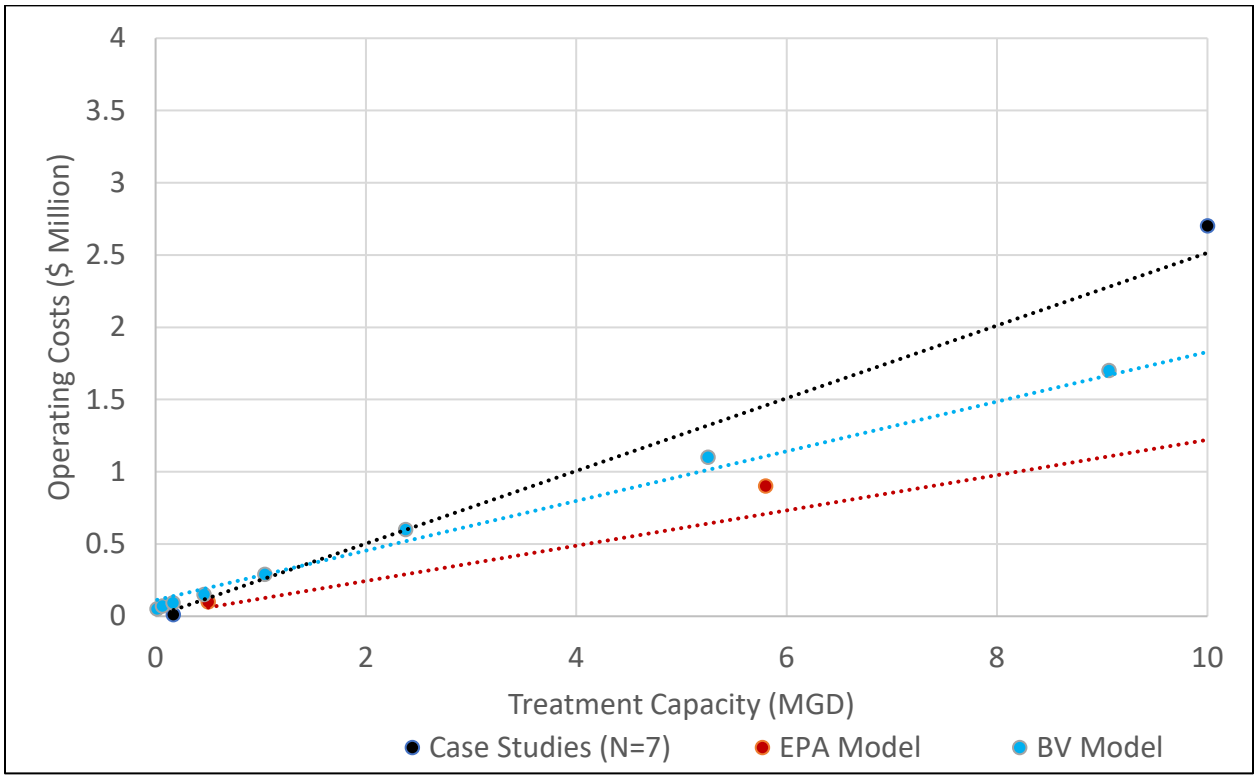


Figure A 24: Comparison of RO Operating Costs for Medium Systems (<10 MGD)

Appendix D

Summary of PFAS Treatment Cost Case Studies

Water System	Treatment Technology	Facility Capacity (MGD)	Facility Capital Cost ($)	Facility Operating Cost ($)	Source
Pease International Tradeport Drinking Water System	GAC	0.4	2	Not Available.	Kleinfelder, 2023; AWWA, 2022b[1]
Merrimack Village District	GAC	2.8	10.9	Not Available.	Kleinfelder, 2023; AWWA, 2023b[1]
Merrimack Village District	GAC	1.8	3.6	Not Available.	Kleinfelder, 2023; AWWA, 2023b[1]
Devens Public Water Supply	GAC	1.25	4.9	Not Available.	Kleinfelder, 2023; AWWA, 2023b[1]
Devens Public Water Supply	GAC	1.12	2.25	Not Available.	Kleinfelder, 2023; AWWA, 2023b[1]
Town of Canton	GAC	2.1	10	Not Available.	Kleinfelder, 2023; AWWA, 2023b[1]
Mashpee Water District	GAC	0.6	2.6	Not Available.	Kleinfelder, 2023; AWWA, 2023b[1]
Mashpee Water District	GAC	0.72	2.3	Not Available.	Kleinfelder, 2023; AWWA, 2023b[1]
Wellesley Water Division	GAC	1.5	6.2	Not Available.	Kleinfelder, 2023; AWWA, 2023b[1]
Anonymous	GAC	0.8	1.5	Not Available.	Kleinfelder, 2023; AWWA, 2023b[1]
Anonymous	GAC	1.2	6	Not Available.	Kleinfelder, 2023; AWWA, 2023b[1]
Anonymous	GAC	2.16	11.8	Not Available.	Kleinfelder, 2023; AWWA, 2023b[1]
Anonymous	GAC	4.03	38	Not Available.	Kleinfelder, 2023; AWWA, 2023b[1]
Anonymous	GAC	0.001	0.2	Not Available.	Kleinfelder, 2023; AWWA, 2023b[1]
Anonymous	GAC	0.03456	0.047	Not Available.	Kleinfelder, 2023; AWWA, 2023b[1]
City of Ames	GAC	15	36.84	1.483	Strand, 2023.
South Central Connecticut Regional Water Authority	GAC	2.5	7.784	0.185	Barger, 2023
South Central Connecticut Regional Water Authority	GAC	2.568	9.816	0.032	Barger, 2023
Pownal Water District	GAC	0.2	2.112	0.110366667	Bennington, 2022
Horsham Water & Sewer District	GAC	0.1	1	0.05	Horsham, 2023b
Horsham Water & Sewer District	GAC	0.5	1	0.05	Horsham, 2023b
Orange County Water District	GAC	62.4	105	Not Available.	Dadakis, 2023
Essential Utilities	GAC	0.32	0.78	0.024811	Essential, 2023
Essential Utilities	GAC	0.58	0.97	0.020054	Essential, 2023
Essential Utilities	GAC	0.36	0.84	0.021642	Essential, 2023
Cape Fear Public Utility Authority	GAC	44	46		AWWA, 2021c
Water System in DE	GAC	30	50.6	2.3	Walczyk, 2023
Water System in ID	GAC	25	105.6	4.8	Walczyk, 2023

Water System	Treatment Technology	Facility Capacity (MGD)	Facility Capital Cost ($)	Facility Operating Cost ($)	Source
Water System in NY	GAC	50	250	11.2	Walczyk, 2023
Hazen & Sawyer Project: Alabama (10 MGD)	GAC	10	9	0.65	Rosenfeldt, 2021
Hazen & Sawyer Project: Alabama (6 MGD)	GAC	6	4.2		Rosenfeldt, 2021
Hazen & Sawyer Project: New Mexico (2 MGD)	GAC	2	4.5	0.088	Rosenfeldt, 2021
Hazen & Sawyer Project: New Mexico (200 gpm)	GAC	0.288	2.7	0.076	Rosenfeldt, 2021
Hazen & Sawyer Project: New York (40 gpm)	GAC	0.0576	1	0.025	Rosenfeldt, 2021
Hazen & Sawyer Project: California (6.2 MGD)	GAC	6.2	15	0.1	Rosenfeldt, 2021
Hazen & Sawyer Project: MAssachusetts (2 MGD)	GAC	2	3	0.045	Rosenfeldt, 2021
Town of Webster, MA	GAC	2.16	18.9	Not Available.	Tighe & Bond, 2022
Town of Webster, MA	GAC	2.88	18.9	Not Available.	Tighe & Bond, 2022
Town of Webster, MA	GAC	2.88	11.45	Not Available.	Tighe & Bond, 2022
Town of Webster, MA	GAC	4.3	35.8	Not Available.	Tighe & Bond, 2022
Lakewood, Colorado	GAC	2.88	3.3	Not Available.	LWD, 2023
Lakewood, Colorado	GAC	4.32	5.1	Not Available.	LWD, 2023
Anonymous	GAC	9.648	26	1	AWWA, 2023b[1]
Anonymous	GAC	44	26.7	2	AWWA, 2023b[1]
Fayetteville Public Works Commission	GAC	39.5	60	7	AWWA, 2023b[1]
Cobb County-Marietta Water Authority	GAC	36	20	1	AWWA, 2023b[1]
Anonymous	GAC	120	240	30	AWWA, 2023b[1]
Anonymous	GAC	9	10	0.6	AWWA, 2023b[1]
Townsend Water	GAC	2.47	21	2	AWWA, 2023b[1]
Town of West Springfield DPW Water Division	GAC	6.5	4	0.25	AWWA, 2023b[1]
Anonymous	GAC	60	0.5	3	AWWA, 2023b[1]
City of Thornton	GAC	50	100	4	AWWA, 2023b[1]
City of Hamilton IL	GAC	1		0.009777	AWWA, 2023b[1]
Anonymous	GAC	5.3	6.251	Not Available.	AWWA, 2023b[1]
Town of Sharon, MA	GAC	2.3	7	0.33	AWWA, 2023b[1]
Anonymous	GAC	80	80	12	AWWA, 2023b[1]

Water System	Treatment Technology	Facility Capacity (MGD)	Facility Capital Cost ($)	Facility Operating Cost ($)	Source
Anonymous	GAC	20	13.8	0.5	AWWA, 2023b[1]
Fayetteville Public Works Commission	GAC	18	20	3	AWWA, 2023b[1]
Norwell Water Department	GAC	0.347	2	0.1	AWWA, 2023b[1]
Anonymous	GAC	60	20	3	AWWA, 2023b[1]
City of Thornton	GAC	20	65	2.6	AWWA, 2023b[1]
Anonymous	GAC	4.0104	13.19	Not Available.	AWWA, 2023b[1]
Anonymous	GAC	70	70	8	AWWA, 2023b[1]
Orange County Utilities	GAC	21.28	35.88	2.166	AWWA, 2023b[1]
Norwell Water Department	GAC	0.288	2	0.1	AWWA, 2023b[1]
Orange County Utilities	GAC	34.56	58.24	1.987	AWWA, 2023b[1]
Merrimack Village District	GAC/IX	1.7	12.42	Not Available.	Kleinfelder, 2023. See note.
Town of Canton	GAC/IX	2.88	25.2	Not Available.	Kleinfelder, 2023. See note.
Frisco, CO Public Works	GAC/IX	1	3	1	AWWA, 2023b[1]
Anonymous	GAC/IX	1.5	12.6		AWWA, 2023b[1]
Norwell Water Department	GAC/IX	1.008	2.6	0.1	AWWA, 2023b[1]
Anonymous	GAC/IX	6	7	Not Available.	AWWA, 2023b[1]
Anonymous	IX	0.576	0.78	Not Available.	Kleinfelder, 2023. See note.
Anonymous	IX	1	4	Not Available.	Kleinfelder, 2023. See note.
City of Ames	IX	15	32.94	2.713	Strand, 2023.
Orange County Water District	IX	4.32	5.805	Not Available.	Dadakis, 2023
Orange County Water District	IX	23.76	24.457	Not Available.	Dadakis, 2023
Orange County Water District	IX	4.32	5.55	Not Available.	Dadakis, 2023
Orange County Water District	IX	4.32	6.194	Not Available.	Dadakis, 2023
Orange County Water District	IX	8.64	25.762	Not Available.	Dadakis, 2023
Orange County Water District	IX	10.8	10.02	Not Available.	Dadakis, 2023
Orange County Water District	IX	25.92	27.7	Not Available.	Dadakis, 2023
Orange County Water District	IX	4.32	6.3	Not Available.	Dadakis, 2023
Orange County Water District	IX	4.32	7.807	Not Available.	Dadakis, 2023

Water System	Treatment Technology	Facility Capacity (MGD)	Facility Capital Cost ($)	Facility Operating Cost ($)	Source
Orange County Water District	IX	8.64	14.2	Not Available.	Dadakis, 2023
Orange County Water District	IX	4.32	4	Not Available.	Dadakis, 2023
Orange County Water District	IX	4.32	4	Not Available.	Dadakis, 2023
Orange County Water District	IX	6.48	8	Not Available.	Dadakis, 2023
Orange County Water District	IX	4.32	4.6	Not Available.	Dadakis, 2023
Orange County Water District	IX	4.32	5.55	Not Available.	Dadakis, 2023
Orange County Water District	IX	4.32	4.431	Not Available.	Dadakis, 2023
Essential Utilities	IX	0.68	4.53	Not Available.	Essential, 2023
Essential Utilities	IX	0.04	0.2	Not Available.	Essential, 2023
Water System in NJ	IX	188	550	25	Walczyk, 2023
Hazen & Sawyer Project: Alabama (10 MGD)	IX	10	13	0.4	Rosenfeldt, 2021
Hazen & Sawyer Project: New Mexico (2 MGD)	IX	2	3.3	0.126	Rosenfeldt, 2021
Hazen & Sawyer Project: New Mexico (200 gpm)	IX	0.288	1	0.072	Rosenfeldt, 2021
Hazen & Sawyer Project: California (6.2 MGD)	IX	6.2	11.1	0.2	Rosenfeldt, 2021
Hazen & Sawyer Project: MAssachusetts (2 MGD)	IX	2	2.25	0.085	Rosenfeldt, 2021
Anonymous	IX	2.9	5.3	0.14	AWWA, 2023b[1]
Anonymous	IX	9.648	25	0.3	AWWA, 2023b[1]
Town of Sharon, MA	IX	1.44	3.8	0.27	AWWA, 2023b[1]
Town of Sharon, MA	IX	1.69	5	Not Available.	AWWA, 2023b[1]
City of Ames	RO	15	63.02	4.065	Strand, 2023.
Brunswick County Public Utilities	RO	36	99	2.9	CDM Smith, 2018.
Hazen & Sawyer Project: Alabama (10 MGD)	RO	10	33	2.7	Rosenfeldt, 2021
Anonymous	RO	0.17	0.85	0.01	AWWA, 2023b[1]
Anonymous	RO	150	706	42	AWWA, 2023b[1]
Anonymous	RO	60	372	20	AWWA, 2023b[1]
Anonymous	RO	250	1.077	60	AWWA, 2023b[1]

[1] For detailed survey data, contact Chris Moody, Regulatory Technical Manager at American Water Works Association.