From: Vicki Liu
Sent: Monday, January 28, 2019 3:33 PM
To: Water <water@cityofmadison.com>
Subject: PFA comment for tomorrows meeting

Hello,

I'd like to comment on two quick points for tomorrows water utility board meeting:

1: Given the levels of PFAs found in Well 15, can this well be shut down until further investigation and action is made in the cleanup of the source of contamination? Our home receives 100% of our water from this well, and the levels (the EPA health levels are loose, which I find great concern with) are so concerning I've contemplated moving and we've only been in this neighborhood a year and a half.

2: What are actionable items we can take to ensure the source of the PFAs will be cleaned? Those responsible need to be held accountable as there are thousands of families and business's using and paying for water that has cancer causing chemicals in it. Given there aren't enough studies to know when side effects//health concerns can take place this needs to be a major priority. (I voice particular concern after seeing the fish we have in our freshwater tank develop tumors and die at a much faster rate then the same fish we had when living on the Northside of Madison for years. The fish we have in RO/Salt water have been unaffected).

Thank you for your time and opening up this platform.

Sincerely,
Vicki Liu

From: Touyeng Xiong
Sent: Monday, January 28, 2019 9:24 PM
To: Water <water@cityofmadison.com>
Subject: Comments to Madison Water Utility Board

City of Madison Water Utility Board,

As a long time resident of the Truax area, and a consumer of drinking water which nearly all comes from Well 15, I am quite concerned for the health of me, my family and neighbors after learning about the contamination of our drinking water with Poly-fluorinated alkyl compounds (PFAS) along with other contaminants. Although this issue has just recently been highlighted, it has been a known problem for some time. People living in the Truax neighborhood already have multiple stressors affecting their lives; not limited to economic, and environmental stressors. Being exposed to these contaminants in addition to already existing issues, would greatly increase our risks of developing health problems. Living in both a at-risk and low-income neighborhood, I believe and suggest that the Water Utility create a comprehensive public engagement plan addressing the PFAS drinking water issues surrounding all wells found to be contaminated with PFAS, particularly focusing on the most at risk neighborhoods such as the Truax Neighborhood.

Thanks for taking my comments into consideration, Please let me know if you have any questions.

Sincerely,
Touyeng

From: Annette Czarnecki
Sent: Monday, February 04, 2019 5:50 PM
To: Water <water@cityofmadison.com>
Subject: water quality

Dear Members of the Madison Water Utility Board,

I have been reading about PFAS in some of our city wells. The more I read about these chemicals the more I am concerned about our drinking water.

The Water Utility should be testing all wells on an ongoing basis. PFAS move quickly through the environment. In theory, all city wells could be affected. I understand PFAS are not regulated by the EPA or WI DNR, but the city needs to do better and get ahead of this situation to determine the scope of the problem. Michigan is already working to identify PFAS contaminated sites.

Thank you for your consideration.

Annette Czarnecki
Madison

CSWAB

CITIZENS FOR SAFE WATER AROUND BADGER
E12629 Weigand's Bay South - Merrimac, WI 53561
Telephone (608) 643-3124
Email: info@cswab.org Website: www.cswab.org
www.facebook.com/cswab.org

February 4, 2019

SENT BY ELECTRONIC MAIL

RE: Madison Utility Board Continued Proactive Approach to PFAS will Benefit City and State

Dear Water Utility Board members:

On behalf of Citizens for Safe Water Around Badger (CSWAB), I would like to thank the City of Madison Water Utility for its proactive response to the detection of low levels of PFAS in certain drinking water wells. By responding promptly to potential public health risks and conducting expanded testing, the Utility has helped assure that community members and agencies have adequate data for well-informed decision making.

The State of Michigan has taken a similar proactive approach. In addition to the very limited testing required by the U.S. EPA, Michigan has begun a statewide initiative to test drinking water from all schools that use well water and community water supplies for PFAS. Michigan is taking this precautionary step of testing these drinking water sources to determine if public health actions are needed.[1]

Here in Wisconsin, CSWAB is pleased to report that Department of Natural Resources recently granted our petitions to establish drinking water Health Advisory Levels for a list of 26 PFAS that have been detected in or pose a threat to the State's groundwater – the source of drinking water for more than two-thirds of its residents.

For all these reasons, comprehensive testing of all water supply wells by the City of Madison for the largest suite of PFAS compounds (30 plus inclusion of the TOP Assay) **will help inform the list of PFAS compounds** that the Wisconsin Division of Health will assess for Health Advisory Levels this year.

A large dataset of results from all Madison area wells **will also help inform the decision to address PFAS as a class of chemicals** which is, in terms of assessing exposure and health effects, the best way to protect public health.[2] The reality is that human exposures are invariably a mixture of PFAS compounds and we need to address total exposure to all PFAS as opposed to the past focus on one substance in isolation.

Thank you for your continued proactive approach to PFAS to the benefit of the City and the State of Wisconsin.

Sincerely,

Laura Olah

Laura Olah, Executive Director
Coordinator, PFAS Community Campaign

Enclosures:

CSWAB Petition for Health Advisory Level HAL Summed Total PFAS August 2018
Wiconsin DNR Letter Granting CSWAB Petition for Summed Total PFAS Jan 2019

[1] Michigan Department of Environmental Quality, PFAS Response: Taking Action, Protective Michigan, *Statewide Testing Initiative*, 2018.

[2] Department of Health and Human Services, National Institutes of Health, National Institute of Environmental Health Sciences, , Hearing on The Federal Role in the Toxic PFAS Chemical Crisis, Testimony before the Senate Committee on Homeland Security and Governmental Affairs, Subcommittee on Federal Spending Oversight and Emergency Management, Linda S. Birnbaum, Ph.D., D.A.B.T., A.T.S., Director, National Institute of Environmental Health Sciences and National Toxicology Program, National Institutes of Health, September 26, 2018.

CSWAB

CITIZENS FOR SAFE WATER AROUND BADGER
E12629 Weigand's Bay South - Merrimac, WI 53561
Telephone (608) 643-3124
Email: info@cswab.org
Website: www.cswab.org
www.facebook.com/cswab.org

August 16, 2018

Steven B. Elmore
Director, Bureau of Drinking Water and Groundwater
Wisconsin Department of Natural Resources
101 S. Webster St., P.O. Box 7921, Madison, WI 53707-7921
Steve.Elmore@wisconsin.gov

SENT BY ELECTRONIC MAIL

RE: Public Petition for Health Advisory Levels for PFAS in Groundwater and Drinking Water with Emphasis on the Tyco/Johnson Controls PFAS site - BRRTS Activity No. 02-38-580694

Dear Director Elmore,

Approximately two-thirds of the people living in Wisconsin rely on groundwater for their drinking water. Adequate supplies of uncontaminated groundwater are crucial to the health of all residents and their families, particularly expectant mothers and newborns.

On behalf of CSWAB, I am writing to request a Health Advisory Level (HAL) for the summed-total concentration of all Per- and Polyfluoroalkyl Substances (PFAS) – including precursors – detected in the State's groundwater and/or having a reasonable probability of entering groundwater such as presence in soils. The persistence and mobility of certain PFAS can lead to large groundwater contaminant plumes extending miles from source areas.

Human health studies have shown that exposure to certain PFAS may affect growth, learning, and behavior of infants and older children, lower a woman's chance of getting pregnant, interfere with the body's natural hormones, increase cholesterol levels, affect the immune system, and increase the risk of cancer. Reference: https://www.atsdr.cdc.gov/pfas/health-effects.html.

There are currently no enforceable federal standards for PFAS in groundwater or drinking water. The U.S. EPA has established a Health Advisory Level for PFOA and PFOS in drinking water however it is not applicable to the complex mixture of PFAS found in Wisconsin's groundwater and affected drinking water wells. Moreover, ATSDR's recently-released draft toxicological profile for perfluoroalkyls provides strong evidence that the current federal HAL is not sufficiently protective.

There is growing evidence that babies, even before they are born, are particularly vulnerable to harm. PFAS in a mother's body can move from her blood into her unborn child and from her breastmilk into her breastfed baby. Therefore we ask that this population in particular be a priority consideration in the development of the requested Wisconsin HAL.

The proposed approach to address PFAS as a mixture is not unusual and is similar to how other groups such as dinitrotoluenes, dioxins, PAHs and PCBs have been assessed and regulated. This approach is consistent with environmental field data which consistently finds PFAS as a mixture of widely varying relative ratios and combinations which, in turn, may shift in response to other factors such as aerobic conditions. This approach is also made necessary by the fact that manufacturers and responsible parties uniformly refuse to disclose PFAS product content and composition, arguing that such information is proprietary.

The most notable industrial PFAS site in Wisconsin is the Tyco Fire Technology Center (Johnson Controls) near Marinette. In March 2018, the Department reported that 36 nearby private drinking water wells were found to be contaminated with PFAS. It is important to note here that this initial testing of residential wells was very limited in scope (6 PFAS analytes) when compared to analysis at the facility where 19 PFAS analytes were both tested and detected. Consequently, the true number of affected homes and analytes present in drinking water supplies is uncertain.

Environmental analysis at the Tyco/Johnson Controls site has detected the following 19 PFAS in groundwater and/or soils:

1. Perfluorobutanesulfonic acid (PFBS)
2. Perfluorohexanesulfonic acid (PFHxS)
3. Perfluoroheptanesulfonic acid (PFHpS)
4. Perfluorooctanoicsulfonic acid (PFOS)
5. Perfluorodecanesulfonic acid (PFDS)
6. Perfluorobutanoic acid (PFBA)
7. Perfluoropentanoic acid (PFPA)
8. Perfluorohexanoic acid (PFHxA)
9. Perfluoroheptanoic acid (PFHpA)
10. Perfluorooctanoic acid (PFOA)
11. Perfluorononanoic acid (PFNA)
12. Perfluorodecanoic acid (PFDA)
13. Perfluoroundecanoic acid (PFUnA)
14. Perfluorododecanoic acid (PFDoA)
15. Perfluorotridecanoic acid (PFTriA)
16. Perfluorotetradecanoic acid (PFTeA)
17. Perfluorooctane sulfonamide (FOSA)
18. 6:2 Fluorotelomer sulfonate (FTS)
19. 8:2 Fluorotelomer sulfonate (FTS)

(Source for above: https://cswab.org/wp-content/uploads/2018/03/Tyco-Ansul-Detects-19-Fluorinated-Compounds-in-Groundwater-2016.pdf)

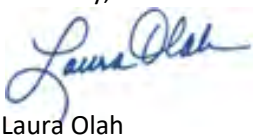
Significant PFAS contamination has also been detected in groundwater at Fort McCoy, General Mitchell 440th, Volk Field, and Truax Field Air National Guard base. It is our understanding that the Department of Defense has not held public meetings in any of these communities to assure that area residents, water utilities, anglers, private well owners, local government and the general public are informed and engaged.

We encourage the Department to write to each of these facilities and encourage public outreach and informational meetings as soon as possible, offering the Department's support and participation in these efforts. We cannot risk failing these communities as occurred at the former Badger Army Ammunition Plant. The public was left in the dark and as a result families were unknowingly exposed to carcinogenic solvents in their drinking water for decades and sadly, there has been a cancer death in each of these families.

An onsite investigation for PFAS at Badger has been proposed however nearby and downgradient community water systems have still not been tested. In fact, the current safety of rural community water supplies throughout Wisconsin is unknown as communities having populations of less than 10,000 are not yet required to test for PFAS and other unregulated drinking water contaminants. A summary of the Department's recommendations and resource needs to address this lack of data is requested.

Thank you in advance for your time and attention. Please do not hesitate to contact me if I may further clarify our requests and purpose.

Sincerely,



Laura Olah
Executive Director

info@cswab.org
608/643-3124

State of Wisconsin
DEPARTMENT OF NATURAL RESOURCES
101 S. Webster Street
Box 7921
Madison WI 53707-7921

Tony Evers, Governor
Preston D. Cole, Secretary
Telephone 608-266-2621
Toll Free 1-888-936-7463
TTY Access via relay - 711

January 17, 2019

Ms. Laura Olah, Executive Director
Citizens for Safe Water Around Badger
E12629 Weigand's Bay South
Merrimac WI 53561

Subject: Public Petition for Health Advisory Levels for PFAS in Groundwater and Drinking Water with Emphasis on the Tyco/Johnson Controls PFAS site - BRRTS Activity No. 02-38-580694

Dear Laura:

Thank you for your August 16, 2018 letter requesting that the department set a Health Advisory Level (HAL) for the summed-total concentration of all Per- and Polyfluoroalkyl Substances (PFAS) – including precursors – detected in the State's groundwater and/or having a reasonable probability of entering groundwater.

As you know, the department submitted a request to the Department of Health Services (DHS) to review two PFAS compounds (PFOS and PFOA), along with 24 other substances on March 2, 2018. DHS will recommend to the department an enforcement standard for each substance, provided that adequate scientific technical information is available.

The department has reviewed your recent petition, along with other PFAS compounds detected in Wisconsin (see the attached table) and has added all of them to the list maintained by the department under Chapter 160 Wis. Stats., thereby granting your petition. Following the established agreement with DHS, the list, including PFAS compounds identified, will be considered for the next submission to DHS under ch. 160, Wis. Stats.

Your petition requests Health Advisory Level (HAL) be established for the summed-total concentration of PFAS. As you know, the United States Environmental Agency (USEPA) has already set federal HALs for PFOA and PFOS both individually and combined at 70 parts per trillion. Wisconsin uses federal HALs when they are available, so Wisconsin already refers to the federal HALs for PFOA and PFOS when evaluating water sampling results. The department will work with DHS to determine the ability to generate HALs for other PFAS based on occurrence, program needs and available toxicological information concurrently with the ch. 160, Wis. Stats. process discussed above. Until DHS reviews the available scientific information, it is unknown whether or not a summed-total concentration is an appropriate approach.

As you are aware, PFAS in groundwater is a national issue, and as such, the department is continuing to work with the USEPA Region 5 and the Association of State Drinking Water Administrators on this complex issue. Some helpful links are included below:

https://www.epa.gov/pfas
https://asdwa.org/pfas

If you have questions regarding this matter please feel free to contact me at (608) 264-9246, or via e-mail at: Steve.Elmore@wisconsin.gov.

Sincerely,

Steven B. Elmore, Director
Bureau of Drinking Water and Groundwater
Wisconsin Department of Natural Resources

Compound	CAS Number
Perfluorooctanoic acid (PFOA)	335-67-1
Perfluoro tridecanoic acid (PFTrDA)	72629-94-8
Perfluoro tetradecanoic acid (PFTeDA)	376-06-7
Perfluoro butanoic acid (PFBA)	375-22-4
Perfluoro pentanoic acid (PFPeA)	2706-90-3
Perfluoro hexanoic acid (PFHxA)	307-24-4
Perfluoro heptanoic acid (PFHpA)	375-85-9
Perfluoro nonanoic acid (PFNA)	375-95-1
Perfluoro decanoic acid (PFDA)	335-76-2
Perfluoro undecanoic acid (PFUdA)	2058-94-8
Perfluoro butanesulfonic acid (PFBS)	375-73-5
Perfluoro hexanesulfonic acid (PFHxS)	355-46-4
Perfluoro octanesulfonic acid (PFOS)	1763-23-1
Perfluoro heptanesulfonic acid (PFHpS)	375-92-8
Perfluoro octanesulfonamide (PFOSA)	754-91-6
Perfluoro-2-methyl-3-oxahexanoic acid (PFPrOPrA)	13252-13-6
Perfluoro dodecanoic acid (PFDoA)	307-55-1
4:2 Fluorotelomer sulfonic acid (4:2_FTS)	757124-72-4
6:2 Fluorotelomer sulfonic acid (6:2_FTS)	27619-97-2
8:2 Fluorotelomer sulfonic acid (8:2_FTS)	39108-34-4
Perfluoro hexadecanoic acid (PFHxDA)	67905-19-5
Perfluoro octandecanoic acid (PFODA)	16517-11-5
Perfluoro decanesulfonic acid (PFDS)	335-77-3
Perfluoro pentanesulfonic acid (PFPeS)	2706-91-4
NaDONA	958445-44-8
F-53B	73606-19-6

From: Susan Pastor
Sent: Tuesday, February 05, 2019 7:50 AM
To: Water Utility Board
Cc: Baldeh, Samba; Grande, Joseph; Water <water@cityofmadison.com>
Subject: Re-sending From Greater Sandburg Neighbors To Water Utility Board

In advance of this afternoon's meeting, I am re-sending this letter from residents (23 and myself) of the Greater Sandburg neighborhood. The rescheduling has allowed for a few more people to sign on. On a personal note, as I have been able to do some research on this issue, I fervently hope we will work to catch up to the broader, precautionary approaches being taken at the community level in Michigan, Vermont, and other places. We need your strong advocacy on behalf of our most precious resource, and that advocacy needs to extend beyond the immediate issue of Well 15. See you this afternoon, and thank you.
--Sue

Dear Madison Water Utility Board Members,

Thank you for putting the issue of Well 15 and PFAS contamination on tonight's agenda. This chance to engage in a public process is especially important to those of us who get the majority of our water from Well 15. The following measures are necessary to address the contamination of Well 15 with per- and poly-fluoroalkyl substances (PFAS) and should be undertaken immediately:

1. We appreciate the new plan to test for a broader range of PFAS contaminants, as the MWU press release reports. We urge that this testing take place as soon as possible, and the results be communicated to the public, as soon as possible.

The tests that showed 39,841 parts per trillion on the base included only two. We need to know more, so we can best understand the health risks and effectively mitigate them.

2. **As a board, formally support Senator Dave Hansen's letter asking the DNR to work with the Wisconsin Department of Health to establish a summed-total PFAS standard for Wisconsin**. The letter can be found at this link. Senator Hansen represents another Wisconsin community affected by PFAS contamination. https://cswab.org/wp-content/uploads/2018/02/Wisconsin-State-Senator-Hansen-letter-to-DNR-PFAS-Enforcement-January-2018.pdf

This is crucial because a precautionary approach is needed to protect the most vulnerable residents (including fetuses and infants)! Other states are continuing to establish lower and more protective standards

than EPA's current health advisory standard for only PFOA/PFOS. In formally supporting Senator Hansen's letter, we ask the Madison Water Utility to urge the DNR to adopt the most precautionary, protective PFAS standards for Madison and all Wisconsin residents, rather than adhering to the EPA advisory, which treats all residents the same.

3. **Convene a Citizens Advisory Panel (CAP) that includes members of vulnerable populations, including those most affected by environmental racism and living adjacent to the base**, to guide the Water Utility and City on selecting an action level/standard for mitigation that is sensitive to the differences among us, a timeline for mitigation that reflects the urgency of this issue, and mitigation options.

4. **As a board, formally ask our federal-level elected representatives to urge the military to begin the testing for PFAS in groundwater off base immediately and establish a plan and clear timeline for cleanup that includes the Starkweather Creek watershed.** The level of just two PFAS contaminants on the base is 39,841 parts per trillion, yet we have no idea of the size or extent of the contaminant plume that is affecting our water.

5. **As a board, formally ask Dane County officials to participate in 2-4 above**. The county owns most of the land generating this hazardous substance (so toxic that the EPA issues a health advisory for the small quantity of 70 parts per trillion) and yet we have heard no public acknowledgement of the problem from county leadership.

Thank you to the Madison Water Utility for all the work so far, and thank you for your attention to this serious issue.

Helen Fitzgerald
Janyce Brickl
Peter Knibbe
Oumar Keita
Jesse and Carolyn Shields
Jon Becker
Delores Robillard
Louis Loui
Anne Dopp
Jeff Lanphear
Barbara Lafferty
Cornelia Clark
Katy Francis Polacek
Jen Jorgensen
Amy Zabransky
Michelle Russell
Kathryn O'Connor
Katherine Pinkston
Charles Spencer
Lindsey Ingram
Heather and Jeff Banschbach
Susan K. Pastor

February 5, 2019

Dear Water Utility Board members:

Firstly, we would like to thank you for including PFAS on today's meeting agenda.

We also thank the City of Madison for requesting on Nov. 28, 2018, with the Water Utility's input, that Truax Air National Guard more fully assess the extent of the PFAS plume emanating offsite from the base and investigate PFAS contamination in ditches leading to Starkweather Creek as well as in surface water and sediments in the creek. We are grateful that city funds were allocated to test PFAS in fish from the creek.

Further, we strongly commend the Water Utility's recent decision, just publicly announced on January 25, to test a larger number of PFAS compounds in Well 15, as we have recommended in the past. This is a wise decision that will help the Water Utility better understand the range and levels of PFAS in this well, make the most informed decisions about public health risks from drinking this water, and choose the best mitigation strategy for this well and other Madison wells in the future.

However, the Madison Water Utility can take further steps to establish itself as a leader in the country in taking the most protective, precautionary actions to protect all of its citizens—especially the most vulnerable people, such as infants and children—from the serious health effects related to PFAS and other contaminant exposures from drinking water.

Precautionary Principle

While there are many scientific, social, methodological, regulatory, economic and other uncertainties and unknowns about PFAS health and environmental risks, there is also a large and growing body of scientific research on PFAS, as well as policy and regulatory knowledge in other states, to guide Madison officials in making proactive decisions to protect public and environmental health.

Moreover, uncertainties and unknowns should not be barriers to taking aggressive actions to protect public health, especially the most vulnerable among us (infants, children). We believe that Madison agencies, policymakers and risk assessors should follow the precautionary principle in regards to PFAS. The core of the precautionary principle is that decisionmakers have social and ethical responsibilities to protect the public from harm even when there are unknowns and uncertainties. Key concepts underlining the principle are very familiar to most of us: "better safe than sorry," "look before you leap," and "first, do no harm."

In addition to assessing the fullest range of PFAS possible at all wells and using the most protective standards, the Water Utility should reduce unknowns and uncertainties by aggressively working to identify sources of PFAS and other contaminants to its wells and taking steps to mitigate them.

The precautionary principle also advocates for as much public participation in decision making as possible in situations involving environmental and public health risks that affect everyone. The Water Utility should initiate and facilitate comprehensive public engagement on PFAS issues, especially with the most at-risk communities, in line with the City of Madison's Racial Equity and Social Justice Initiative.

In this light, thank you for considering our recommendations below:

We ask the Water Utility Board to:
1. Direct the Water Utility to test all Madison drinking water wells as soon as possible for at least 30 PFAS using the lowest available detection limits.
2. Ask the Common Council to create a city (or joint city-county) PFAS Task Force comprised of a wide array of participants, including citizens, to investigate the extent of PFAS problems in our city and county and develop proactive strategies to address them, including setting a local PFAS "action level" for drinking water that would trigger actions to protect residents, especially the most vulnerable people.
3. Direct the Water Utility to facilitate comprehensive public engagement on the PFAS drinking water issues in Madison, including creating a City Resident Advisory Panel for Well 15 and prioritizing targeted engagement with the most at-risk neighborhoods, especially low-income neighborhoods such as Truax.

We also ask the Water Utility Board to formally request in writing that:
4. Madison Metropolitan Sewerage District (MMSD) test PFAS discharged from Truax Air National Guard (ANG) and other sites potentially leaching the compound into city sanitary sewers, as well as PFAS levels in MMSD's wastewater influents/effluents and sewage sludge spread on Dane County farmlands.
5. WI Dept. of Natural Resources (DNR) and Dept. of Health Services prioritize the development of PFAS standards for groundwater/drinking water (and other media) without delay, including a summed-total standard for PFAS compounds.
6. Dane County and the City test groundwater, as well as Starkweather Creek water and sediments, at the two fire-training burn pits on Dane County land for PFAS, VOCs and other contaminants.
7. Truax Air National Guard test volatile organic compounds (VOCs) in addition to PFAS when investigating the groundwater contaminant plume emanating from the base, and if the base is found to be a source, ask the ANG to reimburse the Utility for the costs of the air-stripper currently on the well.
8. Dane County leaders participate openly and publicly in discussions and decisions about what will be done to address PFAS problems, especially on county-owned land (Truax Field).

The attached background document elaborates further on each of the above points.

I'm happy to discuss individual points further and receive input and corrections on them or the background information. PFAS scientific and technical issues are extremely complex and confusing. New and conflicting scientific studies and other relevant information are released nearly every day. We are all learning.

Sincerely,

/s/Maria Powell

Maria Powell, PhD
President, Midwest Environmental Justice Organization

RATIONALE/BACKGROUND FOR MEJO RECOMMENDATIONS

1. All Madison wells should be tested as soon as possible for at least 30 PFAS using the lowest available detection limits.

To date, only five Madison wells (7, 15, 16, 18 and 29) have been tested for PFAS using the lower detection limits. There are a number of other potential PFAS sources throughout the city (e.g., landfills, industries, sewerage treatment, etc.) that could have leached into other Madison wells. PFAS is very mobile and has likely spread from Truax ANG and other sources to wells in addition to Well 15.

Having comprehensive PFAS data from all Madison wells is critical to protecting all Madison residents, especially the most vulnerable (infants, children). This data is also important for making decisions about the appropriate uses, water mixing, and mitigation strategies at various wells. Further, it will help the Water Utility, MMSD, DNR, DHS, and other relevant agencies develop appropriate standards, identify possible sources of PFAS in Madison, and take steps to assure that these sources are mitigated.

2. The Water Utility Board should ask the Common Council to create a city (or joint city-county) PFAS Task Force comprised of a wide array of participants, including citizens, to investigate the extent of PFAS problems in our city and county, identify PFAS sources and develop strategies to address them, including setting a local PFAS "action level" for drinking water that would trigger actions to protect residents, especially the most vulnerable people—infants, children, most at-risk neighborhoods.

The PFAS issues are bigger than the Water Utility's capacities and authority. A city (or joint city-county) PFAS Task Force would be a more appropriate mechanisms to investigate them and develop solutions. A key task of this group would be to develop a local PFAS drinking water "action level" that would serve as a trigger point for decisions to mitigate wells with levels exceeding it or take a range of other actions to prevent harmful exposures, especially to the most vulnerable groups. The level could be revisited and adjusted if deemed appropriate by the Task Force, as new scientific information comes in or state and/or federal PFAS standards are developed.

Why we should <u>not</u> accept EPA's 70 ppt PFOA/PFOS standard for Madison (or wait indefinitely for other standards to be developed by the EPA or Wisconsin):

In developing the 70 ppt standard (for only PFOA and PFOS), EPA was heavily influenced by the Trump Administration, U.S. Department of Defense, as well as industries such as Dupont that produce and use PFAS compounds. The Wisconsin DNR and DHS have also been lagging in addressing PFAS or setting standards for groundwater and other media, and though they have now started to work on developing standards, this will take many months if not years—and after standards are developed it will take years for rules to go through the Legislature and be officially promulgated.

Some states are far ahead of Wisconsin and have chosen not to wait for federal standards. Based on biomonitoring and a growing body of scientific research, a number of scientists and state agencies have concluded that this level is not protective, especially for infants and children, and are taking steps to develop more appropriate standards.

In Minnesota, where agencies have been testing PFAS contaminated groundwater and measuring levels of PFAS in people who drink the water, studies have shown that a pregnant mother can transfer up to 200%

of the PFAS she ingests in drinking water to her developing fetus via the placenta—and that infants take up much higher concentrations from drinking water than adults. Minnesota's biomonitoring and other studies have also shown that when people drink PFAS contaminated water, the levels that build up in their blood serum can be orders of magnitude higher (even thousands of times higher) than levels in water they ingested. Once in the body, PFAS levels increase as the compounds build up in blood and tissues—where they remain for a very long time. An Interstate Technology Regulatory Council (ITRC) fact sheet states that "long-term ingestion of low levels of PFAS (including those below health values) in drinking water may result in exposures substantially higher than in the general population not consuming contaminated drinking water" (highlights added, see citations in the ITRC article).

Based on biomonitoring studies such as the above, along with toxicological and epidemiological studies, several agencies, states, and environmental organizations have proposed standards much lower than EPA's 70 ppt, especially to protect infants/children. The Agency for Toxic Substances and Disease Registry (ATSDR) draft report released in June 2018 (and the agency's November update) suggests limits of 14 ppt for PFOS and 21 ppt for PFOA to protect children. Several states have developed lower standards and more are in the process of doing so. In July 2018, the Vermont Department of Health developed an interim health advisory level of 20 ppt for a summed-total of five PFAS compounds. Last week, the New Jersey Department of Environmental Protection proposed interim groundwater criteria of 10 ppt for PFOS and 10 ppt for PFOA.

Legislators in other states have proposed standards of as low as 5 ppt each for one or both compounds, and Boston University and Harvard scientists have proposed a 1ppt PFOS limit based on their findings of immune system effects in children that affect their responses to vaccines—see here and here. A Natural Resources Defense Council (NRDC) analysis of the ATSDR recommendations proposed levels as low as 3 ppt and 1 ppt for PFOS to prevent immune system effects.

Why would the Water Utility wait? Decisions delayed are health protections denied
At the April 26, 2018 Technical Advisory Committee (TAC) meeting, Dr. Henry Anderson, former Chief Medical Officer for the Wis. Department of Health Services, told Joe Grande that the 70 ppt level would likely drop by at least 50% in the future—but then oddly, at the next meeting, minimized the relevance of ATSDR's draft lower standards for children and advised the Water Utility to "stay the course"—e.g., stick with the 70 ppt EPA standard until agencies develop different standards, which will likely take many years. As I understand it, the committee agreed to this.

If we believe now that the PFAS health advisory levels are likely to go down significantly, and we have abundant evidence that levels much lower than 70 ppt are harmful to infants and children, why would Madison agencies and decisionmakers allow our most vulnerable people to be exposed while we wait indefinitely for EPA to lower the standards or our state to develop state-specific standards? As NRDC stated, in its Sept. 6 comments to the ATSDR, "*Decisions delayed are health protections denied.*"

As I wrote to Joe Grande and Tom Heikkinen on October 8, 2018, after the TAC recommended sticking with the EPA's unprotective 70 ppt health advisory level: "*I would like to live in a city and state that are taking the most proactive, protective approaches to PFAS and other emerging chemicals.*" I would prefer not to have to move elsewhere for this protection. I think most Madison residents would agree.

3. The Water Utility Board should direct the Water Utility to initiate and facilitate comprehensive public engagement on the PFAS drinking water issues, including creating a City Resident Advisory

Panel for Well 15 and prioritize targeted outreach to and engagement with the most at-risk neighborhoods, especially low-income neighborhoods such as the Truax neighborhood.

The low-income Truax neighborhood gets nearly all of its drinking water from Well 15 and is also exposed to PFAS and other contaminants from the military base adjacent to it. People in the Truax neighborhood already live with multiple economic, environmental, cultural and social stressors in addition to exposure to contaminants from Truax Field. These factors act additively and synergistically to increase their risks. This is an important environmental justice issue that should be a priority for the City to address, given its Racial Equity and Social Justice Initiative. To date there has been no engagement with this neighborhood on PFAS or other contamination issues related to Truax Field. People in the Truax apartments should not be expected to know about the Water Utility's websites, sign up for emails to get PFAS info, and to know about and have the ability to attend Water Utility meetings. The Water Utility should actively outreach to this neighborhood through East Madison Community Center leaders and arrange to hold at least one public meeting there about PFAS issues.

4. The Water Utility Board should formally request that the Madison Metropolitan Sewerage District (MMSD) test PFAS discharged from the Truax Air National Guard site and other likely PFAS sources in the city into sanitary sewers, as well as PFAS levels in MMSD's wastewater influents/effluents and sewage sludge spread on Dane County farmlands.

As a January 27 article in the Wisconsin State Journal described, the Marinette Wastewater Utility, using its wastewater permitting authorities, tested PFAS levels dumped into sanitary drains by industries in the city that produce and use fire-fighting foams with PFAS. Levels found in sanitary sewers draining from the industrial sites were significant, as were the influent/effluent levels at the wastewater plant and in sewage sludge from the plant spread on farmland.

If Marinette Wastewater Utility can do this, MMSD can as well. MMSD, like the Marinette Water Treatment plant, has authority to do this through the wastewater permitting program. The District could begin with investigating known PFAS sources such as the Truax Air National Guard base and then address other potential sources in the city as they are identified.

PFAS leaching from old sanitary sewers, sludge lagoons, sludge spread on farmlands, and in surface waters can eventually reach groundwater, so it is in the Water Utility's best interest to have this information. Among other things, it will help MMSD, the Water Utility, and other relevant government agencies identify and mitigate sources of PFAS in Madison.

5. The Water Utility Board should write to the Wisconsin DNR and DHS formally requesting that they prioritize the development of groundwater, surface water, soil, air and other media PFAS standards without delay, including summed-total PFAS standards.

Some Wisconsin residents have already been drinking PFAS for decades. It is imperative that Wisconsin regulatory standards for all media (groundwater/drinking water, surface water, soils, air) be developed without delay, as some other states already have. This is critical to guide Wisconsin municipalities and state agencies on key decisions about safe levels in drinking water and other media, develop appropriate effluent standards, access federal cleanup money, enforce laws, and to guide many other important regulatory and health risk assessments.

The development of summed-total PFAS standards is also critical. Residents are likely drinking many more than two types of PFAS compounds (PFOA and PFOS) right now. To protect people, especially the most vulnerable, it is critical that we develop PFAS standards that consider what people are actually drinking.

Many experts agree with the summed-total PFAS risk assessment approach. The 2014 Helsingør and 2015 Madrid Statements, based on extensive reviews of the scientific literature, provided consensus from more than 200 scientists on the potential for harm associated with the entire class of PFAS. In her testimony before a Senate Committee and Subcommittee on Sept. 26, 2018, Dr. Linda Birnbaum (Director of the National Institute of Environmental Health Sciences and National Toxicology Program of the National Institutes of Health) stated that "Approaching PFAS as a class for assessing exposure and biological impact is the best way to protect public health."

In October 2018 the Conservation Law Foundation sent a petition to the Massachusetts Department of Environmental Protection (DEP) asking the state to adopt a summed-total PFAS standard and as an interim step (while waiting for such a standard to be developed) to adopt Vermont's 20 ppt PFAS standard for five PFAS compounds totaled (see link above).

Here in Wisconsin, in August 2018 Citizens for Safe Water Around Badger (CSWAB) petitioned the DHS to develop a summed-total PFAS standard. In November 2018, Wisconsin's Senator Hansen wrote a letter to DNR and DHS to develop such a standard as well.

6. The Water Utility Board should request that Dane County and the City test groundwater and Starkweather Creek surface water and sediments at and near the two fire-training burn pits on Dane County land for PFAS, VOCs and other contaminants immediately (as soon as weather allows).

Analyses of fire-training burn pits elsewhere have found total PFAS levels in groundwater in the millions of parts-per-trillion. The shallow groundwater at the two Dane County burn pit sites, both former wetlands, is just a few feet down. Starkweather Creek flows around both sites. There is little doubt that PFAS from the repeated use of fire-fighting foams at these sites has leached to the groundwater and creek.

Handing the burn pit investigations off to the National Guard Bureau, as the Mayor and DCRA engineer did on July 31, 2019 is totally unacceptable. This is County-owned land, and the county has authority to test there. The City and County both have authorities to test stormwater drainage from these burn pits.

Further, the City and County both used these burn pits for decades. Both entities have known about the contamination there for at least 30 years but have done nothing to remediate it. Handing this important task to the U.S. military means a long delay in testing and leaves citizens and local decisionmakers in the dark about what is happening, since there appears to be no requirement with this arrangement for public access to the data, public input on the timeline, nature and extent of investigations or options for remedial strategies

7. The Water Utility Board should request that Truax Air National Guard test volatile organic compounds (VOCs) in addition to PFAS when investigating the groundwater contaminant plume emanating from the base. If the base is found to be a source, ask the ANG to reimburse the Utility for the costs of the air-stripper currently on the well.

Well 15 is contaminated with VOCs (PCE levels are near the enforcement standard). After receiving input from the Well 15 Citizen Advisory Panel (CAP) several years ago, a very expensive air-stripper was placed on the well to remove these VOCs. The air-stripper was paid for by the Water Utility (ratepayers). At that time, Truax ANG (which has used VOC solvents including PCE and TCE for decades, and continues to do so), was not considered a source of the VOCs, based on older groundwater modeling.

Updated groundwater modeling done by consultants hired by the Water Utility shows that PFAS can get to Well 15 from the base, and Well 15 PFAS testing to date confirms that it most likely has. If PFAS can get to Well 15 from the base, VOCs can as well.

If further groundwater testing shows that the VOCs from Truax ANG have made it to Well 15, the Air National Guard should be asked to reimburse the Water Utility for the costs of the air stripper (as well as any costs required for PFAS mitigation). Given the financial hardships the Water Utility is facing currently, it seems like the Water Utility would want the responsible party to cover this significant cost. We (the ratepayers) should not be asked to cover mitigation for contamination in our drinking water caused by the U.S. military.

8. The Water Utility Board should formally ask Dane County officials, especially County Executive Joe Parisi, to participate openly and publicly in discussions and decisions about what will be done to address the PFAS problems throughout the county, especially on the land the county owns (such as Truax Field).

Dane County owns most of Truax Field, including the airport, Truax Air National Guard site, fire-training burn pits, former Truax landfill and former Burke Sewage Treatment Plant—all of which are known (or likely) sources of PFAS and other toxic contaminants that are leaching to surface water, Starkweather Creek, and groundwater. We assume the county is engaging in private communications and meetings with city officials about the significant contamination at these sites and their respective liabilities. However, with the exception of a few county supervisors, County Executive Joe Parisi and other county leaders have been totally missing in public discussions to date about the PFAS problem.

In July 2018, MEJO members and interns asked the Lakes & Watershed Commission to create a working group to investigate PFAS and other toxic pollution from Truax Field and its impacts on Starkweather Creek. See MEJO testimonies here (supporting map here), here, here, and here. We requested that this issue be placed on a future Lakes & Watershed agenda, and also asked for more funding in the budget for Starkweather Creek contaminant testing. Along with Laura Olah Executive Director of Citizens for Safe Water Around Badger (CSWAB), we asked the county to help facilitate public meetings on the Truax Field PFAS and other contamination issues.

The Commission dismissed or ignored all of our requests. Our questions and queries to Dane County Executive Joe Parisi have also been met with silence.

This needs to change. Dane County owns most of the highly contaminated Truax Field land including the Air National Guard site. Executive Parisi must publicly take responsibility for addressing the huge, long-term environmental and public health challenges now facing our city and county related to PFAS—and actively engage with the citizens he serves in discussions and decisions about them.

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Evaluation and Management Strategies for Per- and Polyfluoroalkyl Substances (PFASs) in Drinking Water Aquifers: Perspectives from Impacted U.S. Northeast Communities

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Background: Multiple Northeast U.S. communities have discovered per- and polyfluoroalkyl substances (PFASs) in drinking water aquifers in excess of health-based regulatory levels or advisories. Regional stakeholders (consultants, regulators, and others) need technical background and tools to mitigate risks associated with exposure to PFAS-affected groundwater.

Objectives: The aim was to identify challenges faced by stakeholders to extend best practices to other regions experiencing PFAS releases and to establish a framework for research strategies and best management practices.

Methods and Approach: Management challenges were identified during stakeholder engagement events connecting attendees with PFAS experts in focus areas, including fate/transport, toxicology, and regulation. Review of the literature provided perspective on challenges in all focus areas. Publicly available data were used to characterize sources of PFAS impacts in groundwater and conduct a geospatial case study of potential source locations relative to drinking water aquifers in Rhode Island.

Discussion: Challenges in managing PFAS impacts in drinking water arise from the large number of relevant PFASs, unconsolidated information regarding sources, and limited studies on some PFASs. In particular, there is still considerable uncertainty regarding human health impacts of PFASs. Frameworks sequentially evaluating exposure, persistence, and treatability can prioritize PFASs for evaluation of potential human health impacts. A regional case study illustrates how risk-based, geospatial methods can help address knowledge gaps regarding potential sources of PFASs in drinking water aquifers and evaluate risk of exposure.

Conclusion: Lessons learned from stakeholder engagement can assist in developing strategies for management of PFASs in other regions. However, current management practices primarily target a subset of PFASs for which in-depth studies are available. Exposure to less-studied, co-occurring PFASs remains largely unaddressed. Frameworks leveraging the current state of science can be applied toward accelerating this process and reducing exposure to total PFASs in drinking water, even as research regarding health effects continues. https://doi.org/10.1289/EHP2727

Introduction

Per- and polyfluoroalkyl substances (PFASs) exhibit unique chemistry that makes them favorable for use in a wide variety of consumer and industrial products and applications (Kissa 2001). This same chemistry has led to limitations in using traditional environmental chemistry and engineering principles and techniques to understand and manage risks associated with their environmental releases. For example, unlike many neutral organic contaminants, in organisms PFASs are not lipophilic and are known to bind to proteins such as serum albumin (Conder et al. 2008). Additionally, some PFASs are environmentally persistent with no significant natural pathways for complete degradation following release. PFAS chemistry is largely attributable to the strength and low polarizability of the carbon-fluorine covalent bond (Banks et al. 1994; Kissa 2001). PFAS characteristics include thermal stability, chemical stability, surfactant behavior, and stain-resistant properties (Banks et al. 1994; Kissa 2001). Because of these characteristics, PFASs are used in products and applications such as firefighting foams, fluoropolymer manufacturing, stain-resistant coatings, and electroplating. These uses have contributed to their global distribution in organisms and the environment. At the same time, knowledge regarding human health impacts is quite limited, and because of their unique properties, conventional water-treatment techniques do not fully mitigate exposure (DeWitt 2015; Eschauzier et al. 2012; Giesy and Kannan 2001).

Recent studies estimate as many as 3,000 PFASs are now or have been on the global market (Wang et al. 2017). Within this group are perfluoroalkyl substances, which contain an alkyl tail with all carbons bonded to fluorine and which are persistent in the environment (Buck et al. 2011). These perfluoroalkyl substances include PFASs such as perfluorooctanoate (PFOA) and perfluorooctane sulfonate (PFOS), which have been the subject of much of the PFAS research to date. PFOA, PFOS, and their homologues (i.e., shorter and/or longer perfluoroalkyl carboxylates and perfluoroalkyl sulfonates) are often collectively referred to as perfluoroalkyl acids (PFAAs) (Buck et al. 2011). Polyfluoroalkyl substances have at least one perfluoroalkyl moiety (C_nF_{2n+1}) but elsewhere in the structure also contain carbons bonded to hydrogen. These compounds are capable of transformation in the environment (Buck et al. 2011). The terminal degradation products of polyfluoroalkyl substances include PFAAs (e.g., PFOA). So, they are often referred to as precursors and thus still represent a source of recalcitrant PFAAs in the environment (Harding-Marjanovic et al. 2015; Mejia Avendaño and Liu 2015). Examples include fluorotelomer sulfonates, some of which are capable of transforming to PFOA (Harding-Marjanovic et al. 2015).

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There are concerns about the human health impacts of some PFASs, particularly PFOA, PFOS, and other perfluoroalkyl substances. Briefly, in rodent studies, these compounds are known to affect lipid metabolism (e.g., Das et al. 2017) and liver weight (e.g., Loveless et al. 2006), decrease birth weight/increased resorptions (e.g., Lau et al. 2006), delay hind/fore limb phalanges ossification sites in offspring (Lau et al. 2006), delay mammary gland development in offspring (e.g., Tucker et al. 2015), and induce immunosuppression (e.g., DeWitt et al. 2008). Notably, these effects have provided the basis for regulation of PFOA/PFOS. In epidemiological studies conducted by the C8 Science Panel (C8 Science Panel 2017) and others, PFASs have shown positive associations with adverse outcomes, including elevated cholesterol (Frisbee et al. 2010; Nelson et al. 2010), ulcerative colitis (Steenland et al. 2013), thyroid disease (Lopez-Espinosa et al. 2012), testicular and kidney cancer (Barry et al. 2013), childhood adiposity (Braun et al. 2016), decreased duration of breast feeding in infants (Romano et al. 2016), and possibly preeclampsia (Savitz et al. 2012b, 2012a).

Routes of exposure to PFASs include diet (Fromme et al. 2007), dust (Shoeib et al. 2005), and drinking water (Hu et al. 2016). This exposure has led to development of a U.S. Environmental Protection Agency (U.S. EPA) drinking water lifetime health advisory (LHA) for the sum of PFOA and PFOS of 70 ng/L [70 parts per trillion (ppt)] (U.S. EPA 2016b, 2016a), and also regional drinking water standards in the low ppt range (NJDWQI 2016; VTDOH 2016). The U.S. EPA recently completed a national survey of six PFASs in U.S. drinking water that was targeted primarily at large, public, drinking water systems serving more than 10,000 people (U.S. EPA 2012). Studies analyzing these publicly available data (under the Third Unregulated Contaminant Monitoring Rule or UCMR3) have concluded that 6 million U.S. residents are served by systems exceeding the LHA (Hu et al. 2016; U.S. EPA 2012). The survey sampled equal numbers of systems sourced from surface water and groundwater, but approximately 72% of PFAS detections occurred in groundwater (Guelfo and Adamson 2018). Groundwater is the water source for 33% of public supplies in the U.S., and 90% of supplies in rural regions that rely on smaller (i.e., private) wells (USGS 2016). Collectively, health concerns and rates of occurrence highlight the important role of groundwater in human health risks associated with PFAS releases.

Numerous communities in the Northeast U.S. are currently assessing and managing risks due to PFAS-affected groundwater used as drinking water at the private, community, and public scales (e.g., Cape Cod, Massachusetts; Merrimack, New Hampshire; Portsmouth, New Hampshire; Hoosick Falls, New York; Paulsboro, New Jersey; and Bennington, Vermont). In many cases, discovery of these impacts occurred almost simultaneously in a period beginning in 2015, as a result of investigations initiated when residents and regulators learned of potential PFAS sources proximal to drinking water wells. Discovery of PFAS impacts led to a large group of regulators, consultants, analytical laboratories, and responsible parties (herein referred to collectively as stakeholders) with an immediate need to understand and manage risks associated with PFAS-affected groundwater and associated exposure. To determine risks associated with affected groundwater aquifers, stakeholders need information regarding sources, fate and transport pathways, affected receptors, sampling/analytical tools, health-based regulations, and water treatment technologies.

The Brown University Superfund Research Program (SRP) actively engaged with over a thousand stakeholders through PFAS workshops, analytical guidance, and other research translation efforts targeted at communicating the state of the science in the areas of PFAS chemistry, uses, sources, sampling/analysis, fate/transport, remediation, toxicology, regulation, and case studies. These efforts also provided an opportunity to hear diverse perspectives and learn of the key challenges faced by the broader scientific community in managing PFAS impacts. The objectives of this commentary are to (1) compile, critically evaluate, and share research translation findings to rationalize extrapolation of findings to other regions experiencing PFAS releases and (2) provide a framework to guide research and management strategies by prioritizing those PFASs that represent the highest risk of occurrence in treated drinking water. We conclude with an illustrative case study to demonstrate methods that can be used to address knowledge gaps regarding PFASs in drinking water to more effectively evaluate and mitigate risk.

Methods and Approach

The current evaluation utilizes a combination of research translation approaches, review of information in traditional publication outlets, review of publicly available data, and assimilation of select data sources into a limited case study of the risk of PFAS drinking water aquifer impacts in the State of Rhode Island.

Research Translation

To address stakeholder needs in the Northeast, a series of research translation activities were implemented beginning in 2016 to connect stakeholders (primarily from Connecticut, Massachusetts, Maine, New Hampshire, New Jersey, New York, and Vermont) with PFAS experts in various focus areas to communicate relevant aspects of research and current state-of-science (Table S1). Following these efforts, PFAS experts who participated in these events collaborated to document and review the challenges and knowledge gaps that are summarized in the current commentary. A literature review was used to evaluate the knowledge gaps in the framework of the current state-of-science. This review included web searches in the Web of Science database for all English language peer-reviewed articles (e.g., primary data, reviews, editorials) published 1995–present using title and topic search string PFOA OR PFOS OR PFAS in titles and topics from 1995–present ($n = 4{,}249$).

Illustrative Case Study

This commentary presents a geospatial case study to predict risks of PFAS impacts in drinking water aquifers. The approach required an inventory of potential PFAS release sites, which are defined for the case study as facilities that may be associated with the synthesis, use, or disposal of PFASs. We first reviewed peer-reviewed literature and regulatory data to understand potential PFAS source types and associated characteristics (Table 1). Next, we reviewed publicly available regional geospatial coverages and manufacturing directories related to these source types to build a database of facilities in the state of Rhode Island (Table S2). Hard-copy archives of historical manufacturing directories were converted into a digital database using tools described in Berenbaum et al. 2016. Briefly, an open-source data processing tool named GEOREG was created to process the scanned images of Rhode Island manufacturing directories and convert the text into geocoded historical industrial and manufacturing locations from 1950s–present. GEOREG also extracts additional information regarding a facility's name, address, standard industrial classification (SIC) code (i.e., manufacturing type), and number of employees. Resulting data include more than 11,000 unique historical and contemporary manufacturing sites. From this database, we selected only sites that matched the SIC codes (Table S2) and time frame (1960s–present) relevant to PFASs. The application of potential PFAS sources in a geospatial risk evaluation of potential PFAS impacts in Rhode Island groundwater is further discussed in the case study.

We used the characteristics listed in Table 1 to rank PFAS source types according to associated risk for causing groundwater PFAS impacts. First, PFAS source data coverages identified or

Table 1. Groundwater concentrations, compounds, relevant groundwater pathways, and affected receptors resulting from groundwater PFAS source types summarized from peer-reviewed literature and regulatory reports.

Source type	Magnitude of [PFAS] (μg/L)	Max PFAS	PFASs detected	Ground water pathways	Receptors impacted	Ref. cited
PFAS/FP manufacturing	10^{-2}–10^{3}	PFOA	PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFBS, PFHxS, PFOS	VZ to GW Atm SW to GW	DW, GW, SW, B	MDOH 2012; Davis et al. 2007; Bach et al. 2017; Dauchy et al. 2012; Weston Solutions 2009
AFFF use (DoD)[a]	10^{-3}–10^{4}	6:2 FtS	PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFUnA, PFDoA, PFEtS, PFPrS, PFBS, PFPeS, PFHxS, PFHpS, PFOS, PFDS, 4:2 FtS, 6:2 FtS, 8:2 FtS, FHxSA, FOSA, 4:2 FtTAoS, 6:2 FtTAoS, PFBSaAm, PFPeSaAm, PFHxSaAm, PFHxSaAmA	VZ to GW	DW, GW, SW, B	Houtz et al. 2013; McGuire et al. 2014; Schultz et al. 2004; Moody et al. 2003; MDHHS 2016; Hull et al 2017; Moody and Field 1999; Barzen-Hanson and Field 2015; Backe et al. 2013
AFFF use (airport)	10^{-3}–10^{2}	PFOA	PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFBS, PFHxS, PFOS	VZ to GW	DW, GW, SW, B	Ahrens et al. 2015; Awad et al. 2011; Yingling 2016; Antea Group 2011; Delta Consultants 2010; Horsley Witten Group, Inc., 2016
AFFF use (fire training area)[b]	10^{-3}–10^{2}	PFOS	PFBA, PFPeA, PFHxA, PFOA, PFDoA, PFTriA, PFTreA, PFBS, PFHxS, PFOS, EtFASE, MeFASE	VZ to GW	DW, GW, SW	Antea Group 2011; Cape Cod Commission 2016
AFFF use (petroleum)	10^{-3}–10^{1}	PFOS	PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFUnA, PFBS, PFHxS, PFOS, FOSA	VZ to GW	DW, GW	Antea Group 2011
FP coating (e.g. plastics, textiles, metals)	10^{-3}–10^{1}	PFOA	PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFBS, PFHxS, PFHpS, PFOS, FOSA, 6:2 FtS, 8:2 FtS	Not specified	DW, GW	U.S. EPA 2016c; NHDES 2017a
Electronics	10^{-3}–10^{1}	PFOA	PFHpA, PFOA, PFOS	Not specified	DW, GW	Unicorn Mgmt. Consultants 2016
Waste streams (landfills)	10^{-3}–10^{3}	PFBA	PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFBS, PFHxS, PFOS, 6:2 FtS	VA to GW Atm	DW, GW	NHDES 2017a; Weston Solutions 2016; VTDEC 2016; Oliaei et al. 2006; Oliaei et al. 2013
Waste streams (biosolids)	10^{-2}–10^{0}	PFOA	PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFBS, PFHxS, PFOS	VZ to GW	DW, GW, SW, B	Lindstrom et al. 2011
Waste streams (septic systems)	10^{-3}–10^{-2}	PFHxS	PFHxA, PFHpA, PFOA, PFBS, PFHxS, PFOS	VZ to GW	DW, GW	Schaider et al. 2016

Note: PFBA, Perfluorobutanoate; PFPeA, perfluoropentanoate; PFHxA, perfluorohexanoate; PFHpA, perfluoroheptanoate; PFOA, perfluorooctanoate; PFNA, perfluorononanoate; PFDA, perfluorodecanoate; PFUnA, perfluoroundecanoate; PFDoA, perfluorododecanoate; PFTriA, perfluorotridecanoate; PFTreA, perfluorotetradecanoate; PFEtS, perfluoroethane sulfonate; PFPrS, perfluoropropane sulfonate; PFBS, perfluorobutane sulfonate; PFPeS, perfluoropentane sulfonate; PFHxS, perfluorohexane sulfonate; PFHpS, perfluoroheptane sulfonate; PFOS, perfluorooctane sulfonate; PFDS, perfluorodecane sulfonate; 4:2 FtS, 4:2 fluorotelomer sulfonate; 6:2 FtS, 6:2 fluorotelomer sulfonate; 8:2 FtS, 8:2 fluorotelomer sulfonate; FHxSA, perfluorohexane sulfonamide; FOSA, perfluorooctane sulfonamide; 4:2 FtTAoS, 4:2 fluorotelomer thioether amido sulfonate; 6:2 FtTAoS, 6:2 fluorotelomer thioether amido sulfonate; 8:2 fluorotelomerthioether amido sulfonate (8:2 FtTAoS); PFBSaAM, perfluorobutane sulfonamido amine; PFBSaAM, perfluoropentane sulfonamido amine; PFHxSaAm, perfluoropentane sulfonamido amine; PFHxSaAmA, perfluorohexane sulfonamide amino carboxylate; EtFASE, *N*-ethyl perfluoroalkane sulfonamidoethanol; MeFASE, *N*-methyl perfluoroalkane sulfonamidoethanol; VZ, vadose zone; GW, groundwater; SW, surface water; (Atm.) atmospheric deposition and migration through the vadose zone; DW, drinking water, B, biota; DoD, Department of Defense; FP, fluoropolymer.

[a]Recent studies have identified 11 new classes of PFASs comprising 50 individual compounds in AFFF-impacted groundwater from DoD facilities (Barzen-Hanson et al. 2017b); these compounds are not listed here because quantification of their concentrations is not yet available.

[b]Represents fire training areas at municipal or private fire training institutions.

developed for Rhode Island were matched to a source type (i.e., Table 1 Source Type in Table 2). Next, Table 1 was used to determine the number of PFAS compounds known to be present and upper magnitude of PFAS concentrations measured in affected groundwater of each source type. Then, each source type was assigned a risk score of 25, 50, 75, or 100, with the maximum value assigned to those sites yielding highest PFAS groundwater concentrations and number of PFASs (Table 2). The risk score values themselves are arbitrary values used as multipliers along with the duration of operation for each identified facility in calculating a hazard index (HI). A raster of one by one km cells was overlaid on the sources in Rhode Island, and the HI values within a cell were summed (Figure 1). Next, each cell was assigned a groundwater vulnerability index (VI) based on Rhode Island aquifer classifications GAA, GA, GB, and GC (RIDEM 2012). We further divided GAA into groundwater recharge zones and wellhead protection areas. These classifications were given VI values of 100 (GAA, recharge zones), 80 (GAA wellhead protection areas), 60 (GA), 40 (GB) and 20 (GC). Summed HI values were multiplied by VI values to assign a risk index (RI) to each raster cell. Finally, we used a simple universal kriging procedure on the raster cell centroids to smooth RI values across the raster surface and provide some conservative interpolation of the RI values. We used the results to generate risk maps of the Rhode Island region. The risk scores, HI, VI, RI, and groundwater classifications are conceptually described in the discussion.

Results

Key Information Gaps

Sample collection. Investigation of human health and environmental impacts of any compound requires reliable sampling

Table 2. Risk scores utilized for calculation of the PFAS source hazard index (HI).

PFAS source	Upper magnitude (µg/L)	No. PFASs	Risk score	Table 1 source type
DoD facilities	10,000	28	100	AFFF use (DoD)
Chemical manufacturing	1,000	13	100	PFAS/FP manufacturing
Landfills	1,000	11	100	Waste streams (landfills)
Airports	100	28	75	AFFF use (Airports)[a]
Fire training areas	100	28	75	AFFF use (fire training areas)[a]
Petroleum refineries	10	28	75	AFFF use (petroleum refineries)[a]
Textiles	10	13	50	FP coating (plastics, textiles, metals)
Furniture	10	13	50	FP coating (plastics, textiles, metals)
Paper	10	13	50	FP coating (plastics, textiles, metals)
Rubber/plastics	10	13	50	FP coating (plastics, textiles, metals)
Fire Stations	N/A	28	25	N/A[a,b]
Fabricated metal	N/A	11	25	N/A[c]

[a]The number of PFASs reported for this source type was lower in the literature or no data were available (Table 1). A value of 28 was applied because this is the number of PFASs quantified at DoD AFFF-impacted facilities, and it is assumed that an equal number of PFASs may be present at all AFFF-impacted facilities.
[b]There were no data available on groundwater impacts due to fire stations, but fire stations were indicated as a probable source of groundwater impacts during stakeholder engagement. The overall risk score was presumed to be low because many fire stations do not store or use AFFF, and those that do have AFFF do not typically discharge the foams onsite. In personal communications with industry, municipal, and volunteer firefighters, some report that equipment cleaning may occur on site following AFFF use (oral communications, July 2014–July 2017).
[c]There were no data available on groundwater impacts due to electroplating, but data were available on PFASs in waste streams in the chrome plating process (U.S. EPA 2009). These data were used to determine the number of PFASs, and the upper concentration magnitude was the average of the magnitudes from other manufacturing sources.

techniques suitable for concentration levels that may represent health concerns. For PFASs, the challenge arises because of the low regulatory limits (U.S. EPA 2016b, 2016a), their ubiquitous nature (Prevedouros et al. 2006), and their use in the manufacturing process for some types of polytetrafluoroethylene (PTFE), which is a common laboratory material (Kissa 2001). Stakeholders are adopting special precautions (e.g., use of high- or low-density polyethylene containers and silicon tubing) when collecting samples for PFAS analysis (e.g., MassDEP 2017; NHDES 2017b). However, some stakeholders are adopting sampling and analysis protocols that include lists of unallowable items for which the need for prohibition is uncertain or not supported by scientific studies. These protocols include avoiding use of waterproof field notebooks, waterproof clothing, clothing laundered fewer than six times or laundered with fabric softener, cosmetics, and certain sunscreens (MassDEP 2017; NHDES 2017b). There is only limited evidence documenting the presence of PFASs in some of these products (Fujii et al. 2013; Keawmanee et al. 2015), and no cited published data measuring the potential for transfer of PFASs from these materials into samples during collection are available. Therefore, we conclude that the precautions represent an extremely conservative approach to avoid products and materials that include even trace amounts of PFASs. Data are needed to support prioritization of these precautions to avoid unnecessary inconvenience to field sampling personnel. When regulating in the low ppt level, we propose that understanding potential sources of background in samples is also key to differentiating between PFAS-affected drinking water and cross-contamination of samples.

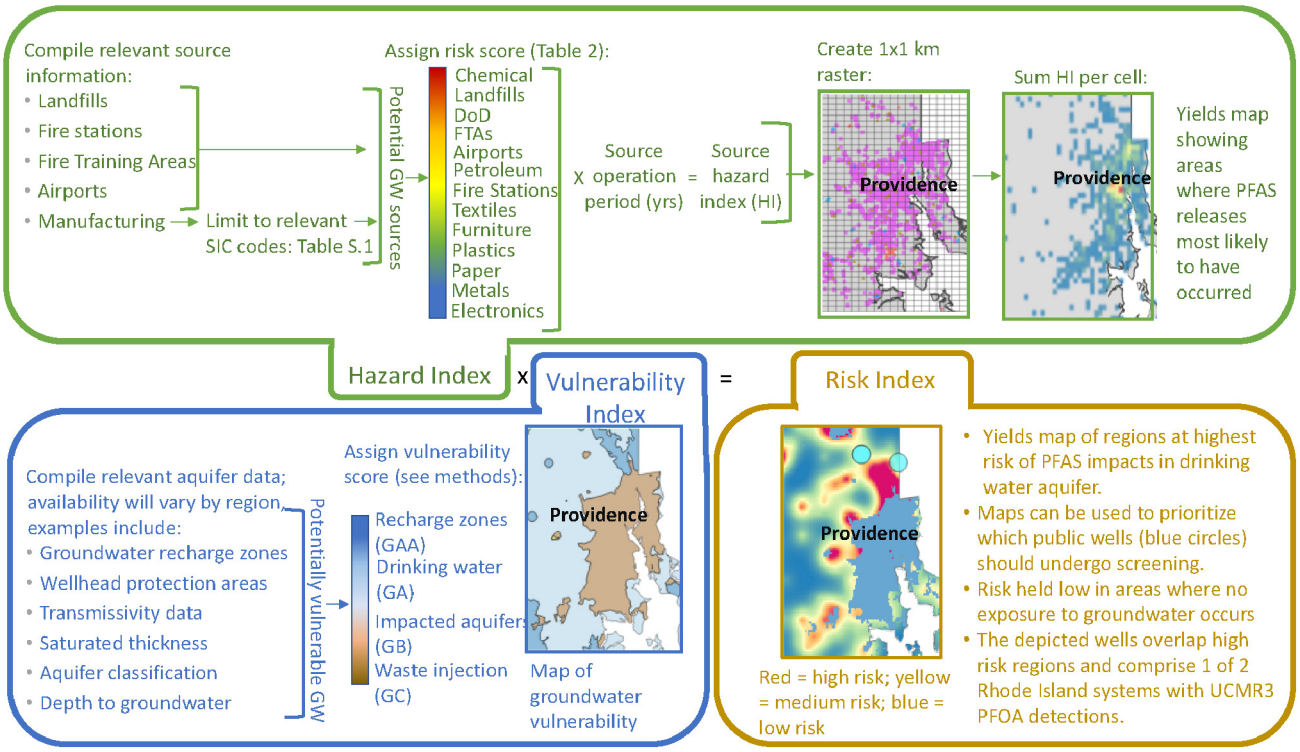


Figure 1. Overview of Rhode Island case study that utilizes a systematic approach to conduct a geospatial risk assessment of potential PFAS impacts in drinking water aquifers. Wells are shown with 1-mile buffers.

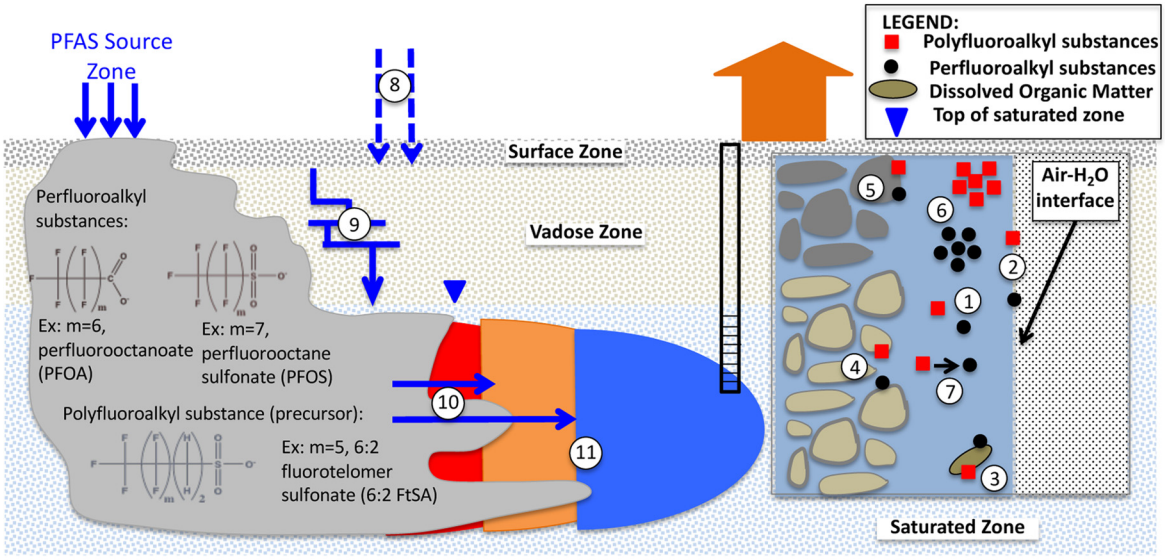
Targeted PFAS analysis. Despite notable progress over the past 20 y in utilizing liquid chromatography tandem mass spectrometry (LC-MS/MS) techniques (and gas chromatographic techniques for volatile PFASs such as fluorotelomer alcohols) for PFAS analysis (e.g., Mahmoud et al. 2009; Moody et al. 2001; Schultz et al. 2004), significant challenges remain. Many obstacles stem from the fact that the complete list of PFASs relevant to environmental and human health exposure scenarios is still unknown and ever increasing as more studies are completed identifying novel PFASs and precursor transformation products (e.g., Barzen-Hanson et al. 2017b). Therefore, though the majority of PFASs are suitable for LC-MS/MS analysis, standards needed to quantify them are not currently available, and it is difficult to keep pace with the increasing number of relevant compounds. It should also be noted that even implementation of targeted LC-MS/MS analysis may represent a challenge in terms of instrument expense, effort, and elimination of PFAS background issues (i.e., from aforementioned laboratory materials such as PTFE) for laboratories that are being required to address the issue. PFAS standards are commercially available (not including special order synthesis) for approximately 70 PFASs, and $\sim 55\%$ of those also have available isotopically labeled versions for use in isotope dilution approaches. It should be noted that in cases where a compound standard is available but a matching labeled standard is not, the labeled version of another PFAS may be used as an internal standard. For example, $^{1,8}0_2$- PFHxS has been used for analysis of PFBS (e.g., Guelfo and Higgins 2013; McGuire et al. 2014). In our view, these challenges make it virtually impossible for any regulatory authority to comprehensively specify which PFASs need to be investigated at a potentially affected site.

Additional analytical tools. Tools are available that can help to characterize the PFAS fraction not quantified during targeted LC-MS/MS analysis. These tools include the total oxidizable precursor (TOP) assay, high-resolution mass spectrometry (HRMS) analysis, particle induced gamma ray emission (PIGE) and adsorbable organofluorine (AOF) analysis. However, these methods are still limited in their availability and ability to quantify concentrations of individual PFASs present at a site. Detailed descriptions of TOP (Houtz and Sedlak 2012), PIGE (Ritter et al. 2017), and AOF (Wagner et al. 2013) are available elsewhere, but briefly, they enable measurement of total precursors, total fluorine, and total organic fluorine, respectively. Coupling TOP, PIGE, or AOF with targeted LC-MS/MS analysis can help researchers understand the total PFAS load present in a sample but does not result in identification of all individual PFASs present. High-resolution mass spectrometry (HRMS) using technology such as quadrupole time of flight generates high mass accuracy data that can be used in identification of unknown compounds (Barzen-Hanson et al. 2017b; Strynar et al. 2015), but quantification of PFASs without standards remains a challenge. TOP has begun to emerge as a commercially used technique, but availability of PIGE, AOF, and HRMS is often limited to noncommercial research laboratories, leaving limited access for regulators and other practitioners who want to implement these tools. During research translation events, stakeholders reported that these challenges prevent them from developing conceptual models of affected sites that include a complete list of PFASs to which environmental and human receptors may be exposed.

Source zone identification. Another key knowledge gap is PFAS source zone identification, which can be illustrated through comparison with the legacy groundwater contaminant, methyl tert-butyl ether (MTBE). MTBE was historically used as a gasoline fuel additive with peak use occurring from 1992–2005 (U.S. EPA 2016d). MTBE has high aqueous solubility (~ 40 g/L), has low soil sorption, and is slow to degrade, leading to the potential for more extensive groundwater plumes relative to other legacy contaminants, such as benzene (Squillace et al. 1996). MTBE had a single primary use (U.S. EPA 2016d), so point sources were commonly facilities or infrastructure associated with retail gasoline supply and distribution for which current and historical information is typically available. Though indirect sources of MTBE groundwater contamination such as atmospheric this deposition are possible, this would primarily contribute to disperse, low-level background concentrations (Squillace et al. 1997) not likely to pose a threat to human health or the environment.

When considering the sources of PFASs and comparing with the example of MTBE, we conclude the following. PFASs also may be highly water soluble with weak soil sorption and exhibit recalcitrance to natural degradation, leading to the potential for large groundwater plumes. However, unlike MTBE, there are many relevant PFASs and diverse products and applications with which they are associated (Table 1). Additionally, although PFASs may be associated with a particular process or product, such as textiles manufacturing, it cannot be concluded that all products and manufacturers in a relevant industrial category utilized PFASs. Finally, due to regulatory limits in the low ppt range, indirect sources of PFASs (e.g., groundwater impacts due to leaching of atmospheric deposition or land application of composts and wastewater biosolids) have led to environmentally relevant groundwater impacts (Lindstrom et al. 2011; Shin et al. 2011) and cannot be discounted as important PFAS sources. Assembling information on all potential PFAS sources in a particular region is further complicated because information on current and historical locations of both direct and indirect sources is often missing or unconsolidated. For stakeholders who need to identify sources of known PFAS impacts or to design targeted screening programs to assess if releases have occurred, unconsolidated source data may lead to inefficient or untargeted sampling plans, a failure to identify all sources relevant to a particular release, or the inability to determine a source, thereby increasing time required to reduce risks to public health and the environment.

Subsurface fate and transport. Although studies have investigated PFAS fate and transport, significant knowledge gaps remain (Figure 2). There are knowledge gaps in the general areas of PFAS composition [Figure 2 (1)] partitioning [Figure 2 (2–6)] transformation [Figure 2 (7)], and the influence of site hydrogeology and geochemistry [Figure 2 (8–11)]. An understanding of fate and transport requires knowledge of both compound-specific (e.g., sorption, transformation) and site-specific (e.g., geology, geochemistry) factors (Fetter 1999). PFAAs are recalcitrant (Prevedouros et al. 2006), so the primary compound-specific factors that need to be understood are sorption and potential for generation from precursors. Polyfluoroalkyl substances (i.e., precursors) can transform (Harding-Marjanovic et al. 2015; Mejia Avendaño and Liu 2015; Weiner et al. 2013), so in addition to sorption, knowledge of transformation rates and pathways (e.g., intermediate products) is required. These processes rely on compound-specific properties, such as sorbate structure, but are also influenced by site-specific properties, such as sorbent type, solution chemistry, and cocontaminants (Guelfo and Higgins 2013; Higgins and Luthy 2006; Weber et al. 2017). Further, PFAS distribution at the field scale also depends on subsurface hydrogeologic conditions, including groundwater flow direction, velocity, and influence of heterogeneous geology (Fetter 1999). Finally, user-friendly groundwater modeling tools for legacy contaminants have been developed to assist stakeholders with decision points, such as site prioritization, monitoring plans and duration, and design of aquifer remediation alternatives (Aziz and Newell 2000; Newell et al. 1996), but comparable tools



Description	Key Knowledge Gaps
(1) Dissolved PFAS (see Table 1 for current state of science)	Total PFAS composition at key points (surface, vadose, saturated zones) at PFAS-impacted facilities
(2) PFAS air-water partitioning[1,2]	Impact of PFAS water-air surface activity on partitioning in unsaturated media
(3) PFAS sorption to dissolved organic matter	Potential for PFAS to partition to DOM and exhibit facilitated transport, similar to other contaminants
(4) PFAS sorption to solids[3-10]	Potential for PFAS sorption hysteresis, polyfluoroalkyl sorption, impacts of soil/solution chemistry on polyfluoroalkyl sorption
(5) PFAS sorption to solids altered by co-contaminants or remedial activity[10--13]	Impact of co-contaminants, remediation on sorption/desorption of polyfluoroalkyl substances
(6) PFAS micelle/hemimicelle formation	Potential formation of PFAS micelles or hemimicelles as concentrations increase and saturation decreases
(7) Precursor biotransformation to perfluoroalkyl substances[14-16]	Transformation pathways, applicability of laboratory precursor transformation rates at the site level
(8) Infiltration	Impacts of infiltration and resulting changes in saturation on PFAS transport through the surface and vadose zones
(9) Vadose zone hydrogeology	Influence of fractures, root zone, low permeability zones, or capillary zone on PFAS migration to the saturated zone
(10) Saturated zone hydrogeology	Impact of variable hydraulic conductivity or other heterogeneous subsurface conditions on saturated PFAS transport
(11) Transition from reducing to oxic conditions[7,10]	Potential for redox conditions caused by co-contaminants or their degradation products (e.g. CH_4) to influence zones of precursor transformation

Figure 2. Conceptual model of micro and macroscale PFAS fate/transport processes and associated knowledge gaps. Superscripted numbers refer to the following references: [1]Kissa 2001; [2]Banks et al. 1994; [3]Higgins and Luthy 2006; [4]Liu and Lee 2007; [5]Liu and Lee 2005; [6]Ferrey et al. 2012; [7]Ololade et al. 2016; [8]Tang et al. 2010; [9]Barzen-Hanson et al. 2017a; [10]Weber et al. 2017; [11]Guelfo and Higgins 2013; [12]McKenzie et al. 2015; [13]McKenzie et al. 2016; [14]Harding-Marjanovic et al. 2015; [15]Mejia Avendaño and Liu 2015; [16]Weiner et al. 2013.

are currently not available for PFASs. In our opinion, these challenges in understanding PFAS fate and transport limit the ability to identify at-risk receptors, understand the probability for a source to affect those receptors, and to help identify potential sources of newly discovered groundwater releases.

PFAS toxicology and use in regulation. Other efforts have summarized the current state of science regarding PFAS toxicology (ASTDR 2015; DeWitt et al. 2009, 2015; Lau et al. 2007; Negri et al. 2017) and epidemiology (Bach et al. 2015, 2016, Chang et al. 2014; Negri et al. 2017; Steenland et al. 2010). In our view, results of toxicology studies highlight several challenges related to understanding health effects of PFASs. First, studies address the toxicology of only a subset of PFASs. Outside of PFOA/PFOS, toxicology studies are available for other PFAAs, such as PFBA mouse studies (Das et al. 2008; Foreman et al. 2009), PFBS rat studies (Lieder et al. 2009a, 2009b), a PFHxA rat study (Loveless et al. 2009), PFNA mouse studies (Das et al. 2015; Fang et al. 2008), a PFNA rat study (Feng et al. 2010), and a PFHxS rat study (Butenhoff et al. 2009). Studies are also generally limited for polyfluoroalkyl compounds (Buck 2015), especially recent replacement products, although some data are available for PFOA replacement products GenX (Beekman et al.

2016; Caverly Rae et al. 2015; Gannon et al. 2016) and ADONA (Gordon 2011). Second, the mechanism of action for PFAA-associated toxicity is not well understood, although peroxisome proliferator-activated receptor alpha (PPARα) activation is often implicated (Das et al. 2015; Lau et al. 2007). Third, although animal studies are useful in elucidating target organs, there are notable differences in how humans and animals interact with PFAAs. For example, PFAAs are documented to have half-lives on the order of years in humans (Li et al. 2018; Worley et al. 2017) but only hours to days in laboratory animals (Lau et al. 2007). Long half-lives indicate human serum concentrations will remain elevated, suggesting toxicity in humans may persist even after environmental PFAS levels decrease. Further, applying PPARα activation in animal studies to humans is complicated by several species differences that are well described elsewhere (Post et al. 2017). Fourth, the potential for synergistic toxicity is not well characterized, despite human exposure to PFAS mixtures. Two related *in vitro* studies found that mixtures of 2–4 PFASs yielded additive, not synergistic, activation of murine PPARα (Carr et al. 2013; Wolf et al. 2014). However, as noted, questions remain regarding the role of PPARα and the applicability of animal PPARα studies to humans. Additionally, *in vitro* studies are unable to capture pharmacokinetics alterations that may lead to synergistic toxicity.

Knowledge gaps in toxicology pose challenges for regulators and other stakeholders tasked with managing PFAS releases. Although data are available for PFOS and PFOA, there is still a lack of consensus regarding which toxicological end point and subpopulations should be targeted in development of drinking water standards. This lack of consensus can be illustrated through comparison of the U.S. EPA LHAs (U.S. EPA 2016b, 2016a) to standards developed for PFOA in New Jersey (NJDWQI 2016) and Vermont (VTDOH 2016). Drinking water quality standards (DWQS) are generally calculated as follows (e.g., U.S. EPA 2016b, 2016a):

$$DWQS = \frac{RfD * BW}{DWI} * RSC \quad [1]$$

where RfD is the reference dose (i.e., the maximum daily dose for which no adverse health effects are expected to occur), BW is the body weight, DWI is the drinking water ingestion rate, and RSC is the relative source contribution, or proportion of PFAS exposure from drinking water. The LHA (U.S. EPA 2016b, 2016a) and DWQS differ between New Jersey (NJDWQI 2016) and Vermont (VTDOH 2016) in part because of key differences in the values used to calculate these standards (Table 3). In standards development, values for DWI may change, depending on the subpopulation considered (e.g., adults, children), and, in the case of PFOA agencies, differ on which rates are deemed adequately protective: lactating women (U.S. EPA), BW adjusted rate for the first year of life (Vermont), or adult water intake (New Jersey). Table 3 also illustrates that there is debate regarding the most appropriate RfD. Specifically, the NJDWQI believes that the EPA failed to consider more sensitive end points, such as the liver and immune effects for which the NJ RfD is considered protective (NJDWQI 2016). The NJDWQI also expressed concerns, which we share, regarding low-dose findings, such as lack of repetition, nonmonotonic data (delay in phalanges ossification and mammary gland development), and unknown clinical significance (mammary gland development without disruption of lactation, delay in phalanges ossification without malformations, mild reductions in immune factors without increase incident of infection) (NJDWQI 2016).

Although all approaches result in standards in the low ppt range, these variations lead to different interpretations of what would be considered an affected drinking water system. For example, the U.S. EPA UCMR3 efforts sampled ~5,000 public drinking water systems in the U.S. for six PFASs, including PFOS, PFOA, and PFNA (U.S. EPA 2016e). The number of systems in the data set that would be considered problematic based on the U.S. EPA LHAs more than doubles if the New Jersey or Vermont standards are applied to the data set (Guelfo and Adamson 2018). Additionally, different standards may be applied to drinking water of adjacent communities separated by state lines. Such is the case in New York and Vermont, raising questions about why a community in one state may continue to drink groundwater that would be considered unsafe by an adjacent community across a state line.

Groundwater remediation. Conventional treatment techniques are ineffective for removal or destruction of the full suite of PFASs present in affected water (e.g., Rahman et al. 2014; Schultz et al. 2006). For example, processes relying on *in situ* chemical oxidation cannot fully destroy all PFASs but can enhance oxidation of precursors to end point PFAAs (Houtz and Sedlak 2012). Additionally, they may destroy perfluoroalkyl carboxylates (e.g., PFOA) under some conditions but are ineffective at degrading perfluoroalkyl sulfonates (e.g., PFOS) (Bruton and Sedlak 2017; Park et al. 2016). Filtration with granular activated carbon (GAC) and anion exchange resins (AER) have been shown to remove PFOS and PFOA but may not be as effective for treatment of short chain PFAAs and precursors (Appleman et al. 2013; Xiao et al. 2017; Yu et al. 2009; Zaggia et al. 2016). Further, filtration does not achieve compound destruction, so additional treatment or disposal of spent media is required. There has been some success at the bench and pilot scale using advanced oxidation processes that rely on electrochemical or plasma-based techniques to destroy PFASs in extracted, affected groundwater (Chaplin 2014; Stratton et al. 2017). Despite significant progress, these techniques are generally not ready for full-scale implementation, and key concerns include potential treatment-rate limitations and energy requirements. Last, design of any treatment technique may also need to account for cocontaminants (e.g., hydrocarbon constituents) that may be present in some aquifers (McGuire et al. 2014).

Despite knowledge gaps, filtration with GAC is a common technique used to address PFAS in affected drinking water systems (e.g., Damon 2016; NYDEC 2016; Weston & Sampson 2016). Although effective for treatment of PFASs currently targeted for regulation (i.e., PFOS, PFOA), these systems are not optimized for removal of the full suite of PFASs present in some affected groundwater. In our view, this suggests exposure to PFASs for which toxicological outcomes are not yet fully understood may be ongoing. In the event that water quality standards are developed for additional PFASs, we also point to the potential scenario that sites formerly remediated for PFOS/PFOA will need to be revisited for treatment of PFASs not previously considered. To minimize these

Table 3. Values used in development of PFOA advisories and standards and associated maximum recommended levels in drinking water.

Agency	Advisory or standard (ng/L)	RfD (mg/kg – day)	DWI/BW (L/kg – day)	RSC	Toxicological end point	Reference
USEPA	70	2.E-05	0.054	0.2	delay in phalanges ossification, mice	U.S. EPA 2016b, 2016a
NJDWQI	14	2.E-06	0.029	0.2	Hepatoxicity, mice	NJDWQI 2016
VTDOH	20	2.E-05	0.175	0.2	delay in phalanges ossification, mice	VTDOH 2016

Note: NJDWQI, New Jersey Drinking Water Quality Institute; VTDOH, Vermont Department of Health; DWI, drinking water ingestion rate; BW, body weight; RSC, relative source contribution.

potential risks, we agree with others that it may be necessary to implement combined remedies or treatment trains (Crimi et al. 2017; Kucharzyk et al. 2017) and argue that these approaches should target removal or destruction of total PFASs present.

Discussion

Often during site assessment, evaluation of levels of exposure and potential human health consequences are of paramount concern (U.S. EPA 1989). In our view, even when areas such as occurrence, fate/transport, and remediation are well understood, the health consequences of PFASs will remain uncertain. Although the volume of research in both toxicology and human health studies has increased markedly in the last decade, firm conclusions relating individual PFASs to specific health outcomes have remained elusive. As noted, a wide range of potential links between PFAS exposure and health outcomes have been reported (e.g., ASTDR 2015; Steenland et al. 2010), but the uncertainties remain substantial. As one extends that interest into other legacy PFASs and particularly into the newer generation of PFASs, the empirical evidence guiding interpretation of health effects declines substantially and is virtually absent for many of the compounds, suggesting a need for strategies to prioritize PFASs for further study.

PFAS Framework

Frameworks have been developed in previous studies to evaluate large groups of compounds and prioritize those that should be targeted for further research on factors such as analysis, occurrence, fate and transport, and treatability (de Voogt et al. 2009; Howard and Muir 2010, 2011, 2013; Kumar and Xagoraraki 2010; Strempel et al. 2012), but these approaches often rely at least in part on toxicity information, the limitations of which have already been discussed for PFASs. Nevertheless, we propose that additional criteria used in these approaches, such as occurrence, persistence, and treatability, might be coupled with evaluation of exposure into a framework to guide future research and inform best management practices (Figure 3). Despite challenges outlined herein, progress has been made in understanding drinking water occurrence, persistence, and treatability, such that it is possible to begin identifying PFASs that should be targeted for further study. That is particularly the case for evaluation of exposure (Step 1), and here we present a limited case study illustrating a risk-based evaluation of the potential for PFAS exposure in drinking water due to presence of potential source zones. Last, we note that the body of publicly available information regarding aspects such as compound persistence and toxicity continues to grow as part of legislative efforts, such as the European Union's Registration, Evaluation, Authorization and Restriction of Chemicals (REACH) (ECHA 2017a). REACH requires industry to identify and communicate risks associated with substances they use (ECHA 2017b), creating a valuable resource that can be applied toward compound prioritization using this framework.

Geospatial Evaluation of PFAS Exposure Risk

As discussed, there are challenges in identifying sources of and exposure to PFAS groundwater impacts on a regional scale. We present a case study of potential PFAS groundwater impacts in the state of Rhode Island to illustrate methods that can be used to overcome these challenges. Previous studies have assessed regional risks of degraded groundwater quality due to other classes of contaminants by compiling information on potential sources of contamination (e.g., population-dense regions, landfills, gas stations) and comparing their location with groundwater that is vulnerable to impacts (Babiker et al. 2005; Rahman 2008). Groundwater vulnerability may be evaluated through hydrogeologic characteristics, such as depth to groundwater and transmissivity (Gemitzi et al. 2006; Wang et al. 2012). A similar approach can be adopted for PFASs by compiling data on potential PFAS sources and groundwater vulnerability and applying a risk-based system to evaluate the potential for PFAS impacts (Figure 1).

As described in the Methods section, risk scores were assigned to each PFAS source type (Table 2). This score was based on the fact that not all PFAS sources are likely to be associated with the same (or any) severity of groundwater effects. A limitation to the calculation of the risk scores is that the number of PFASs detected at a given type of site may be a product of the limited PFASs that were analyzed. For example, many precursors could be present that have never been investigated. Additionally, some source types. such as electronics facilities and metal plating. had limited or no groundwater data from affected sites, so the upper magnitude of concentrations may not be representative. Nevertheless, the data are considered to be representative in terms of understanding the different source types relative to each other. Further, the resulting source ranking that reflects Department of Defense (DoD), chemicals manufacturing, and landfills as the highest risk sites is consistent with literature and regulatory reports in terms of capturing release types that are known to have caused significant drinking water aquifer PFAS impacts (Davis et al. 2007; Moody et al. 2003; NH DHHS 2016; Oliaei et al. 2013). Risk scores were used to calculate HI values (see Methods) which, when mapped, depict areas where PFAS releases are most likely to have occurred (Figure 1).

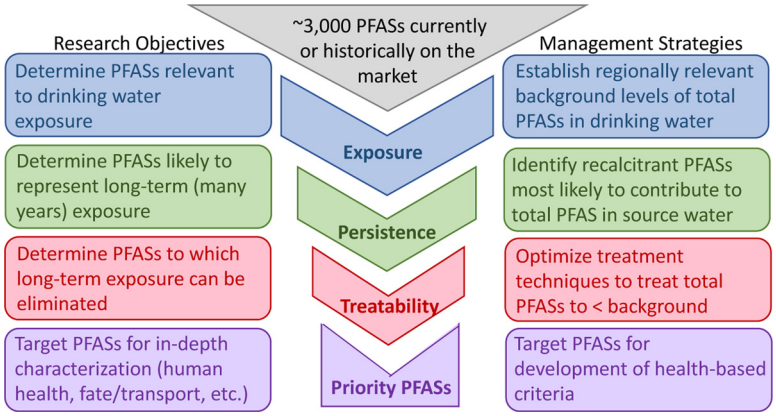


Figure 3. Framework for research and management strategies that prioritize PFASs based on highest risk of exposure in drinking water. It should be noted that here exposure refers only to drinking water; other routes of ingestion, such as food, are not considered.

The HI, or likelihood of PFAS release, should not be used as the sole indicator for potential aquifer impacts and subsequent exposure because releases that occur must also be transported through the subsurface to the saturated zone in order to affect aquifers, and certain hydrogeologic conditions make this more likely (i.e., higher groundwater vulnerability) in some regions (Fetter 1999). Rhode Island aquifer classifications already represent an evaluation of groundwater vulnerability. The State of Rhode Island classifications are based on locations of shallow zones of recharge to deeper aquifers and wellhead protection areas (GAA), drinking water aquifers evaluated based on transmissivity and saturated aquifer thickness (GA), aquifers presumed or known to be degraded (GB), and groundwater where waste injection is permitted (GC) (RIDEM 2012). VI values based on these classifications (see Methods) define the risk of environmental releases affecting usable groundwater when such events occur (Figure 1). When the HI and VI are combined to calculate RI, the highest RI values represent areas at highest risk for PFAS impacts in drinking water and subsequent exposure (Figure 1).

Results of the evaluation of groundwater PFAS impacts in Rhode Island reveal high HI values centered around regions with a high density of former manufacturing facilities that are situated in the population-dense region of Providence. Although there is a high likelihood of a PFAS release having occurred in this region, the RI values near Providence are low due to depressed VI values. Aquifers in the immediate vicinity of Providence are classified as GB, indicating that there is no use of the groundwater (RIDEM 2012), so there should be no drinking water exposure to PFAS impacts present. The highest RI values occurred in more rural regions with a lower density of sources (i.e., lower HI) but higher VI values due to proximity to groundwater recharge areas and drinking water aquifers. RI maps can be applied towards understanding potential sources of known PFAS groundwater impacts or in prioritizing drinking water wells that should be targeted in sampling programs, with the ultimate goal of understanding and mitigating risks associated with PFAS exposure. In Rhode Island, the majority of wells in high RI regions are private or small community wells, which were not screened as part of U.S. EPA UCMR3 screening efforts. Notably, two PFOA detections were discovered in Rhode Island as part of UCMR3, with concentrations of $20-81$ ng/L. The geospatial evaluation of PFASs in Rhode Island aquifers found that wells in both of these systems (some systems are sourced from multiple wells) have overlap with areas of high RI (Figure 1). Finally, it should be noted that this approach represents a limited case study for illustrative purposes, and efforts to further ground truth, refine the geospatial approach, and characterize which (if any) PFASs are present are part of ongoing research.

Conclusions

In summary, interactions with stakeholders from affected communities in the Northeast U.S. have identified a number of key knowledge gaps in several areas, including sampling and analysis, fate and transport, toxicology, regulation, and water treatment. An important result is a lack of consensus regarding management and regulation of PFASs in drinking water. Both laboratory and epidemiological studies support the potential for negative health outcomes due to PFAS exposure. In response, water quality regulations for PFASs are starting to emerge, but these regulations primarily apply to PFOS/PFOA. Regulatory levels are based solely on extrapolation from mechanistic studies in animal models and incorporate substantial uncertainty factors as a margin of safety. We conclude that these recently recommended standards and advisories for select PFASs should not be interpreted as indicating that health will be adversely affected if levels are exceeded. Rather, in explaining the health implications of elevated levels of PFASs in water sources to various stakeholders, it is important to be clear that knowledge is limited, in most cases severely so, and that declaration of safe or harmful levels of contamination is not possible.

Despite this uncertainty, we believe that it is important not to be complacent about human exposure to PFASs via drinking water, and strategies are needed to begin addressing water quality impacts even as research is ongoing. Therefore, we conclude that understanding knowledge gaps will help to guide investigation, management, and mitigation of specific releases, and that the framework developed here can be used to facilitate broader strategies for research and management focused on total PFASs in drinking water. The latter will help accelerate the process of mitigating exposure to PFASs for which detailed studies are lacking.

Results presented herein suggest that it is possible to begin implementing a comprehensive strategy towards PFAS management despite the considerable gaps in current knowledge, particularly regarding toxicity. In particular, this work compiles an already a large body of evidence related to potential PFAS sources and occurrence in groundwater (Table 1) that can be applied toward understanding exposure. We illustrated this by performing a risk-based, geospatial case study of potential PFAS source zones in Rhode Island drinking water aquifers. When compared with limited groundwater aquifer results, high-risk zones identified in the geospatial evaluation were proximal to drinking water wells with detectable PFAS concentrations (Figure 1). Further, new regulations such as REACH have led to increased sharing of industry data related to compound behavior in the environment, and this sharing helps build connections between research in academia, industry, and government. In our view, growing such partnerships facilitates effective management of chemical use and release. This commentary focuses on streamlined research strategies and best management practices for PFASs in drinking water, and this focus could be extended to evaluate other routes of exposure. Similar approaches might also be applied to other complex mixtures of aqueous contaminants with the overall effect of leveraging the current state of the science towards understanding drinking water impacts and reducing risks to human health.

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From: Vicki Liu
Sent: Tuesday, February 05, 2019 10:59 AM
To: Water <water@cityofmadison.com>
Subject: Two Studies to consider in Safe PFA Levels

Hello,

I recently spoke to a leading researcher (Alissa Cordner), who published an insightful study about "drinking water advisory levels", as well as a referral to a well documented study on the adverse affects of PFA exposure by Philippe Grandjean and Dick Clapp done in 2015. In this particular study, the outcome recommended 1ppt as a safere exposure level for PFA's in drinking water (please look at the bottom of page 153 to see the effects on childrens health).

I think these two papers have enough scientific data to show the current "safety levels" we following, are not safe at all. I implore us to consider these studies findings. I'd hate to see our city be listed in the future as one that didnt take the right action when we could have to ensure safety for our citizens.

Thank you for your consideration and time.

Vicki Liu

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ARTICLE

Guideline levels for PFOA and PFOS in drinking water: the role of scientific uncertainty, risk assessment decisions, and social factors

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Abstract

Communities across the U.S. are discovering drinking water contaminated by perfluoroalkyl and polyfluoroalkyl substances (PFAS) and determining appropriate actions. There are currently no federal PFAS drinking water standards despite widespread drinking water contamination, ubiquitous population-level exposure, and toxicological and epidemiological evidence of adverse health effects. Absent federal PFAS standards, multiple U.S. states have developed their own health-based water guideline levels to guide decisions about contaminated site cleanup and drinking water surveillance and treatment. We examined perfluorooctanoic acid (PFOA) and perfluorooctane sulfonate (PFOS) water guideline levels developed by the U.S. Environmental Protection Agency (EPA) and state agencies to protect people drinking the water, and summarized how and why these levels differ. We referenced documents and tables released in June 2018 by the Interstate Technology and Regulatory Council (ITRC) to identify states that have drinking water and groundwater guideline levels for PFOA and/or PFOS that differ from EPA's health advisories (HAs). We also gathered assessment documents from state websites and contacted state environmental and health agencies to identify and confirm current guidelines. Seven states have developed their own water guideline levels for PFOA and/or PFOS ranging from 13 to 1000 ng/L, compared to EPA's HA of 70 ng/L for both compounds individually or combined. We find that the development of PFAS guideline levels via exposure and hazard assessment decisions is influenced by multiple scientific, technical, and social factors, including managing scientific uncertainty, technical decisions and capacity, and social, political, and economic influences from involved stakeholders. Assessments by multiple states and academic scientists suggest that EPA's HA is not sufficiently protective. The ability of states to develop their own guideline levels and standards provides diverse risk assessment approaches as models for other state and federal regulators, while a sufficiently protective, scientifically sound, and enforceable federal standard would provide more consistent protection.

Keywords Drinking water · Emerging contaminants · Exposure assessment · Perfluorinated chemicals · PFAS · Risk assessment

Introduction

The mobility, persistence, and widespread use of perfluoroalkyl and polyfluoroalkyl substances (PFAS) have resulted in drinking water contamination globally. PFAS were found in the drinking water of more than 16 million Americans in 33 states [1], and a recent analysis indicates that PFAS-contaminated drinking water is much more widespread than previously reported [2]. Surprisingly, despite this widespread contamination [3], ubiquitous exposure [4], and toxicological and epidemiological evidence of health effects [5–7], there are no federal drinking water standards for any PFAS. Instead of a standard, in 2016 the U.S. Environmental Protection Agency (EPA) released a non-enforceable lifetime health advisory (HA)

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of 70 ng/L for perfluorooctanoic acid (PFOA) and perfluorooctane sulfonate (PFOS), individually or combined. Without an enforceable standard, public water systems (PWSs) are not required to routinely test for PFAS or to treat water exceeding EPA HAs, and so no complete assessment of the prevalence of PFAS in U.S. drinking water exists.

In the absence of federal standards, seven U.S. states have adopted or proposed their own health-based drinking water guideline levels or standards for PFOA and/or PFOS, ranging from 13 to 1000 ng/L. There are important regulatory distinctions between terms such as guidelines, advisories, and standards. For this paper, we use "drinking water guideline levels" as a general term to refer to any risk-based water concentration intended to protect from health effects associated with drinking water consumption, along with more precise terms that are used by individual state or federal agencies, including "health advisory level," "maximum contaminant level," or "protective concentration level." (Tables 1 and 2 use the specific term associated with each agency's guideline.)

In this perspective, we compare PFOA and PFOS drinking water guideline levels developed by EPA and seven states, and summarize how and why these levels differ. We aim to provide a useful overview of a rapidly changing regulatory field, identify common factors and decisions that influence guideline development, and examine the importance of social factors. We used tables released by the Interstate Technology and Regulatory Council (ITRC) in June 2018 [8] to identify states with drinking water and groundwater guideline levels for PFOA and/or PFOS that differ from EPA's HAs. These documents serve as a resource for regulatory personnel addressing PFAS contamination and are updated regularly by a team of environmental professionals. We also contacted state health and environmental agencies to identify and confirm current guideline levels. For all guidelines, we reviewed publicly available risk assessment documents and toxicological summaries prepared by regulatory agencies.

We find that the development of PFOA and PFOS guideline levels is influenced by many scientific, technical, and social factors and decisions including: agency management of scientific uncertainty; an evolving understanding of PFAS health effects; decisions about toxicological endpoints and exposure parameters; and the influence of various stakeholders, including regulated industries and affected communities. We document the rationale used by states to develop guideline levels that differ from those set by EPA. Several states have established guideline levels below EPA's HA, suggesting that some regulators and scientists view EPA's approach as not sufficiently protective.

Perfluoroalkyl and polyfluoroalkyl substances: growing concerns

PFAS as a class include an estimated 4730 human-made and commercially available chemicals, polymers, and mixtures containing chains of fluorinated carbon atoms that are widely used in industrial processes and consumer goods [9]. It is not currently possible to accurately track the use of PFAS individually or as a class in the U.S. because companies can claim production volume data as confidential business information and not disclose it publicly or to EPA. Two PFAS are the most well-known and widely studied. PFOA—previously used to manufacture polytetrafluoroethylene (PTFE) for non-stick coatings such as Teflon™, added as an ingredient in firefighting foams, and created as a byproduct of many other chemical processes—was first used to manufacture commercial products in 1949. U.S. manufacturer DuPont began studying PFOA's toxicological and exposure concerns starting in the 1960s [10]. PFOS, previously used in fabric protectors such as Scotchgard™, firefighting foam, and semiconductor devices, has been produced since the 1940s. U.S. manufacturer 3M started measuring fluorine levels in blood samples from workers in the 1970s [11]. In 1997, 3M detected PFOS in workers' blood serum and in samples from U.S. blood banks, intended to represent a control population, and several studies in following years confirmed widespread exposure in the U.S. population [12]. In 2000, 3M announced that it would voluntarily phase out all production of PFOS due to regulatory pressure and concerns over liability [13]. In 2006, following an EPA investigation, eight U.S. chemical manufacturers agreed to phase out all production and use of PFOA and related compounds by 2015 [14]. PFOA and PFOS, both considered long-chain PFAS (perfluorocarboxylic acids with eight or more carbon atoms or perfluorosulfonic acids with six or more carbon atoms [15]), are no longer produced in the U.S., but manufacturing continues in other parts of the world [16] and replacement PFAS are widely used despite growing concerns about persistence, exposure, and toxicity [14, 17–21].

PFAS are important and widespread drinking water contaminants because they are highly persistent, mobile in groundwater, and bioaccumulative [22]. PFAS contamination is often linked to industrial releases, waste disposal and landfill sites, military fire training areas, airports, and other sites where PFAS-containing aqueous film-forming foams (AFFFs) are used to extinguish flammable liquid fuel fires or for firefighter training [1]. Over twenty-five U.S. communities have contaminated water due to releases from manufacturing or industrial waste sites [23], and the Department of Defense (DoD) has identified 401 current or former military sites with known or suspected PFAS contamination, including 126 sites with PFOA or PFOS

Table 1 PFOA drinking water guideline levels

	Advisory level	Critical effect study	Toxicological endpoint	Reference dose	Uncertainty factors	Target population	Water ingestion rate	RSC
U.S. EPA[a], 2016, Health Advisory Level [35]	70 ng/L	Lau et al. [49]	Developmental	20 ng/kg- day	Total = 300 Intraspecies 10, Interspecies 3, LOAEL to NOAEL 10	Lactating woman	0.054 L/kg-day	20%
Alaska DEC[b], 2016, Groundwater cleanup level [89]	400 ng/L	Lau et al. [49]	Developmental	20 ng/kg- day	Total = 300 Intraspecies 10, Interspecies 3, LOAEL to NOAEL 10	Child (0–6 years) residential	0.78 L/day, 15 kg body weight (b.w.)	100%
Maine DEP[b], 2016, Remedial action guideline [90, 91]	130 ng/L	Six studies combined [49, 92–94]	Liver	6 ng/kg-day	Total = 300 Intraspecies 10, Interspecies 3, Database 10	Adult	2 L/day, 70 kg b.w.	60%
Minnesota DOH, 2017, Non-cancer health-based level [95]	35 ng/L	Lau et al. [49]	Developmental	18 ng/kg- day	Total = 300 Intraspecies 10, Interspecies 3, LOAEL to NOAEL 3, Database 3	Infant exposure via breastmilk for 1 year, from mother chronically exposed via drinking water	Derived from internal serum concentrations based on 95% water intake rates and upper percentile breastmilk intake rates	50%
New Jersey DEP, 2017, Maximum contaminant level (recommended) [45]	14 ng/L	Loveless et al. [96]	Liver	2 ng/kg-day	Total = 300 Intraspecies = 10, Interspecies 3, Database 10	Adult	2 L/day, 70 kg b.w.	20%
North Carolina DENR[b], 2012, Interim maximum allowable concentration (proposed) [54]	1000 ng/L	Butenhoff et al. [97]	Liver	N/A	Total = 30 Intraspecies 10, Interspecies 3	Adult	2 L/day, 70 kg b.w.	20%
Texas CEQ[b], 2017, Protective concentration level [81]	290 ng/L	Macon et al. [50]	Mammary Gland	15 ng/kg- day	Total = 300 Intraspecies 10, LOAEL to NOAEL 30	Child (0–6 years) residential	0.64 L/day, 15 kg b.w.	100%
Vermont[a] DEC/DOH, 2016, Primary groundwater enforcement standard [98]	20 ng/L	Lau et al. [49]	Developmental	20 ng/kg- day	Total = 300 Intraspecies 10, Interspecies 3, LOAEL to NOAEL 10	Infant (0–1 year)	0.175 L/kg-day	20%

Note: Adapted from ITRC [8]

CEQ Commission on Environmental Quality, *DEC* Department of Environmental Conservation, *DENR* Department of Environment and Natural Resources (note that NC DENR is now NC DEQ), *DEP* Department of Environmental Protection, *DEQ* Department of Environmental Quality, *DOH* Department of Health, *RSC* Relative Source Contribution

[a]Applies to PFOA and PFOS individually, as well as the sum of PFOA and PFOS

[b]Alaska, Maine, North Carolina, and Texas follow the EPA's HA for public and/or private drinking water

Table 2 PFOS drinking water guideline levels

	Advisory level	Critical effect study	Toxicological endpoint	Reference dose	Uncertainty factors	Target population	Water ingestion rate	RSC
U.S. EPA[a] Office of Water, 2016, Health Advisory Level [35]	70 ng/L	Luebker et al. [99]	Reduced pup body weight	20 ng/kg- day	Total = 30 Interspecies 3, Intraspecies 10	Lactating women	0.054 L/kg-day	20%
Alaska DEC[b], 2016, Groundwater cleanup level [89]	400 ng/L	Luebker et al. [99]	Reduced pup body weight	20 ng/kg- day	Total = 30 Interspecies 10, Intraspecies 3	Child (0–6 years) residential, non-cancer	0.78 L/day, 15 kg b.w.	100%
Maine DEP[b], 2016, Remedial action guideline [91, 100]	560 ng/L	Seacat et al. [101]	Thyroid effects	80 ng/kg- day	Total = 30 Interspecies 3, Intraspecies 10	Adult	2 L/day, 70 kg b.w.	20%
Minnesota DOH, 2017, non-cancer health-based value [102]	27 ng/L	Luebker et al. [99]	Reduced pup body weight	5.1 ng/kg- day	Total = 100 Interspecies 3, Intraspecies 10, Database 3	Lifetime based on internal serum concentration	Derived from internal serum concentrations based on 95% water intake rates and upper percentile breastmilk intake rates	50%
New Jersey DEP, 2017, Maximum contaminant level, draft [103]	13 ng/L	Dong et al. [104]	Immune response	1.8 ng/kg- day	Total= 30 Interspecies 3, Sensitive subpopulations 10	Adult	2 L/day, 70 kg b.w.	20%
Texas CEQ[b], 2017, Protective concentration level [81]	560 ng/L	Zeng et al. [105]	Hippocampus synapse structure	20 ng/kg- day	Total = 100 LOAEL to NOAEL 10, Intraspecies 10	Child (0–6 years) residential	0.64 L/day, 15 kg b.w.	100%
Vermont[a] DEC/DOH, 2016, Primary groundwater enforcement standard [98]	20 ng/L	Luebker et al. [99]	Reduced pup body weight	20 ng/kg- day	Total = 30 Interspecies 3, Intraspecies 10	Infant (0–1 year)	0.175 L/kg-day	20%

Note: Adapted from ITRC [8]

CEQ Commission on Environmental Quality, *DEC* Department of Environmental Conservation, *DEP* Department of Environmental Protection, *DEQ* Department of Environmental Quality, *DOH* Department of Health, *RSC* Relative Source Contribution

[a]Applies to PFOA and PFOS individually, as well as the sum of PFOA and PFOS

[b]Alaska, Maine, and Texas follow the EPA's HA for public and/or private drinking water

levels above EPA's HA, mostly related to AFFF use [24]. In addition to PFOA and PFOS, 57 classes of PFAS have been identified in AFFF and/or AFFF-contaminated groundwater, containing over 240 individual compounds, many of which are poorly characterized in terms of toxicity and environmental fate and transport [25]. Surveillance for PFAS is difficult because of the large number of compounds, many of which lack analytical standards.

Concern about health effects from PFAS is high because of widespread exposure and documented toxicity. Biomonitoring data from the U.S. Centers for Disease Control and Prevention's National Health and Nutrition Examination Survey (NHANES), a representative sample of U.S. residents, for 12 PFAS from 1999 to 2014 found four PFAS in the serum of nearly all people tested [4, 26]. These PFAS remain widely detected, although population serum levels have generally declined, especially for PFOS, following the phase-outs of U.S. production [26]. An epidemiological study, funded by a DuPont lawsuit settlement, of 69,000 people in the Mid-Ohio Valley who drank water contaminated with at least 50 ng/L of PFOA for at least one year linked PFOA exposure to high cholesterol, ulcerative colitis, thyroid disease, testicular and kidney cancers, and pregnancy-induced hypertension [6]. Other health effects associated with PFOA and several other PFAS based on epidemiological evidence include decreased vaccine response, liver damage, and decreased birth weight [27, 28]. In animal studies, PFAS have shown a variety of toxicological effects including liver toxicity, suppressed immune function, altered mammary gland development, obesity, and cancer [7, 22]. There is concordance between some of the endpoints identified in studies of animals and humans, most notably suppression of the immune system [29]. While there are sufficient data for risk assessment of PFOA, PFOS, and several other PFAS, most PFAS detected in drinking water lack sufficient data for risk characterization [22, 28].

Drinking water regulation

Public drinking water supplies (PWSs) in the U.S. are regulated under the Safe Drinking Water Act (SDWA), which specifies that EPA is responsible for establishing testing requirements and standards, while states have primary authority to implement and enforce these standards. The SDWA currently regulates over 90 chemical, biological, and radiological contaminants [30]. For most listed contaminants, EPA establishes both a Maximum Contaminant Level Goal (MCLG), a non-enforceable guideline below which no adverse health effects are expected, and a Maximum Contaminant Level (MCL), an enforceable standard for PWSs set as close as feasible to the MCLG while accounting for availability of treatment technologies and cost. PWSs must test for regulated contaminants, which can reveal previously unrecognized contamination, and take any needed action to address violations. Amendments to the SDWA in 1996 removed a requirement for EPA to periodically establish new MCLs and created a more extensive review process, and few additional contaminants have been regulated since 1996 [31]. Private drinking water sources are not regulated under the SDWA. Other laws like the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA, also known as Superfund) and the Clean Water Act govern groundwater and surface water quality, including responses to contaminated water at industrial sites. States often develop health-based water guidelines to support decisions at these sites, including response to contamination in private wells.

EPA has not set MCLs for any PFAS, though they recently announced their intention to "initiate steps to evaluate the need for a maximum contaminant level (MCL) for PFOA and PFOS" [32]. In an unusual move that reflects the political demand for a federal MCL, 25 U.S. Senators signed a letter urging EPA to develop an MCL for PFAS [33]. Establishment of an MCL would increase EPA's authority to address PFAS contamination under the Superfund program [33].

The SDWA also requires EPA to consider additional contaminants for regulation. Every five years, EPA must publish a Candidate Contaminant List (CCL) of contaminants being considered for future standards based on health concerns, prevalence in PWSs, and meaningful opportunities for exposure reduction [34]. No MCLs have been developed for contaminants from the CCL since the SDWA 1996 Amendments were enacted [31]. PFOS and PFOA were added to the third CCL in 2009 and were carried forward to the fourth CCL in 2016. To inform this process, every five years EPA must also develop a list of up to 30 contaminants under the Unregulated Contaminant Monitoring Rule (UCMR) program for which PWSs are required to test on a short-term basis to establish their prevalence. In the third cycle (UCMR3; 2013–2015), six PFAS were analyzed by all large PWSs (serving >10,000 customers) and 800 smaller PWSs [3]. EPA decided not to include any PFAS in UCMR4 (2018–2020).

Under the SDWA, EPA can establish HAs for contaminants without MCLs as guidance for federal, state, and local officials. HAs are intended to represent levels of exposure unlikely to cause adverse health effects, considering both cancer and non-cancer endpoints, and can represent specific durations of exposure (one-day, 10-day, or lifetime). Federal HAs and state guidance values can guide response at contaminated sites if drinking

water is affected but do not require PWSs to proactively monitor for these contaminants. In 2016, EPA issued HAs for lifetime PFOA and PFOS exposure [3, 35].

Individual states can also establish their own guidelines and regulations, including MCLs, for drinking water contaminants that are not regulated at the federal level, or they can develop stricter guidelines for contaminants with a federal MCL. There is precedent for states to develop drinking water MCLs for contaminants that do not have federal MCLs (e.g., perchlorate in Massachusetts and methyl tertiary-butyl ether in California) or to develop MCLs that are more stringent than EPA's (e.g., several volatile solvents in New Jersey and California) [36–38]. These state standards and guidelines may apply to PWSs or be used as screening or cleanup levels at contaminated sites (e.g., sites with contaminated groundwater or drinking water). However, some states are precluded by state law from developing their own guidelines or standards, and other states may lack the resources to do so. For instance, Pennsylvania identified lack of funding, technical expertize, and occurrence data as challenges in setting a state standard for PFOA and PFOS [39].

Variation in PFOA and PFOS drinking water guideline levels

In the absence of federal MCLs, multiple states have proposed or adopted drinking water guidelines or standards for PFOA and/or PFOS (Fig. 1). The first PFOA guideline level of 150,000 ng/L was developed in West Virginia in 2002 in response to PFOA-contaminated drinking water near a DuPont facility. In 2006, EPA issued a screening level of 500 ng/L for PFOA for West Virginia sites contaminated by DuPont [40]. In 2009, EPA developed provisional, short-term HAs of 400 ng/L for PFOA and 200 ng/L for PFOS in response to a contaminated site in Alabama. Around the same time, states such as Minnesota and New Jersey developed PFOA guidelines and standards that were lower than the EPA's short-term HA. In 2016, EPA issued a lifetime HA of 70 ng/L for PFOA and PFOS individually or combined [3, 35]. Shortly after, Vermont and Minnesota, building off the EPA's risk assessments, developed state guideline levels that were lower than the EPA HAs. In 2017, New Jersey recommended MCLs of 14 ng/L for PFOA and 13 ng/L for PFOS, which, if adopted, would be the first standards to require surveillance by PWSs for PFOA and PFOS, as well as being the lowest guideline levels in the U.S.

We analyzed fifteen current or proposed water guidelines or standards for PFOA or PFOS that are the most recent guidelines for the EPA and each state: EPA's PFOA and PFOS HAs, seven state guidelines for PFOA, and six state guidelines for PFOS (Tables 1 and 2). Some states (e.g., New Jersey and North Carolina) have

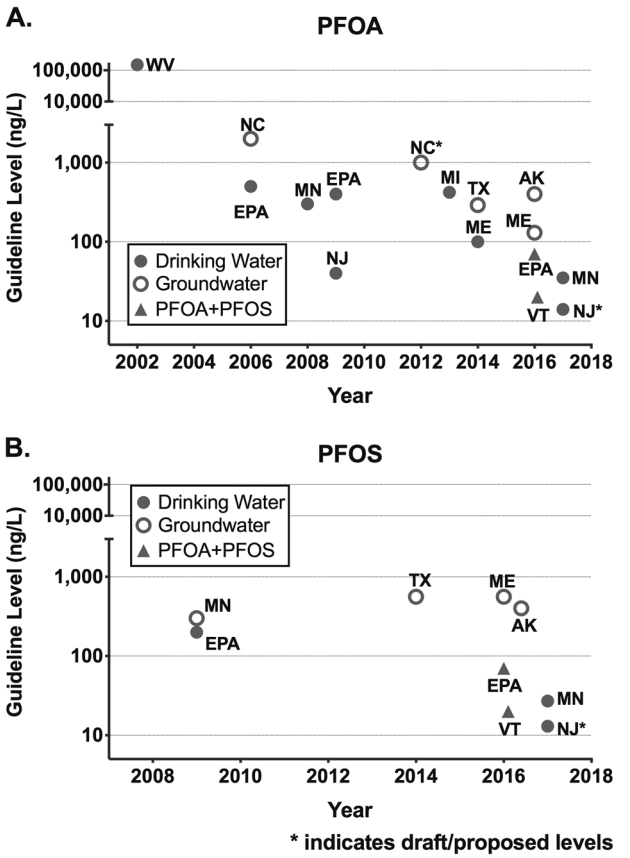

Fig. 1 Timeline of Select PFOA and PFOS Drinking Water Guideline Levels. (**a**) PFOA and (**b**) PFOS water guideline levels have decreased over time. Several states have developed guidelines for PFOA or PFOS individually (circles), while Vermont (VT) and EPA have guidelines that apply to PFOA and PFOS individually or combined (triangles). PFOA and PFOS water guidelines can apply to different water types such as public drinking water (closed circles) or groundwater, e.g., at contaminated sites (open circles)

older adopted guidelines, as well as newer proposed guidelines that have not yet been formally adopted; in these cases, we analyzed the more recent, proposed guidelines. Some guideline levels apply to individual chemicals, while others are based on the sum of multiple PFAS. For example, the EPA HA applies to PFOA and PFOS combined, and the Connecticut, Massachusetts, and Vermont guidelines refer to the sum of PFOA, PFOS, and three other PFAS [41–43]. Eight states (Colorado, Delaware, Massachusetts, Minnesota, New Jersey, North Carolina, and Texas) have developed guideline levels for PFAS other than PFOA and PFOS. Many other states follow EPA's 70 ng/L HA level and are not included in our analysis or shown in the Figure or Tables.

The most recent proposed state guideline levels for PFOA vary by a factor of 70, from 14 ng/L (New Jersey) to 1000 ng/L (North Carolina; Table 1). For PFOS, the seven guidelines vary by a factor of 43, from 13 ng/L (New Jersey) to 560 ng/L (Maine and Texas; Table 2).

Alaska, Maine, and Texas follow EPA's HA for public and/or private drinking water supplies but have developed higher guideline levels for other contaminated water and site remediation intended to be protective of drinking water exposures from groundwater at those contaminated sites.

PFOA and PFOS health-based risk assessment

Comparing the risk assessments developed by states and EPA to derive these guideline levels highlights the scientific uncertainty and assumptions that underlie these decisions. Tables 1 and 2 summarize critical components of each assessment: toxicological endpoint, critical study, uncertainty factors, target population, and exposure parameters.

Toxicological and dose-response assessments

Risk assessment is used to develop health-based guideline levels. Scientists first review toxicological, epidemiological, and mode of action studies to identify the critical effect, the most sensitive adverse endpoint that is considered relevant to humans. Four of the eight guideline levels for PFOA are based on developmental effects, three are based on liver toxicity, and one is based on mammary gland development effects. Of the seven guideline levels for PFOS, four are based on reduced pup body weight, one is based on thyroid effects, one is based on suppressed immune response, and one is based on developmental neurotoxicity. New Jersey's recommended PFOS MCL, the lowest in the country, is the only assessment to use immune response as the critical endpoint.

The critical effect serves as the starting point for deriving a point of departure (POD), the point on the dose-response curve to which uncertainty factors (UFs) are applied, such as a No Observed Adverse Effect Level (NOAEL) or Lowest Observed Adverse Effect Level (LOAEL). In PFAS assessments, toxicokinetic adjustments were made to account for slower excretion of PFOA and PFOS in humans compared to animals, either by calculating a Human Equivalent Dose based on doses used in animal studies (most states and EPA) or by converting serum levels based on animal studies into serum levels in humans (New Jersey). This is a particularly important consideration for PFAS because of substantial variation in PFAS toxicokinetics among humans and test animals [44]. There are also sex-specific and species-specific differences in the excretion rates of PFAS. For example, PFOA has a very short half-life in female rats (4–6 h) due to rapid excretion [44], which makes the female rat a poor model for studying chronic or developmental effects of PFOA exposure since it is unlikely to reach a steady-state level when administered on a daily basis.

After a POD is derived, UFs are applied to the POD for non-cancer endpoints to estimate a reference dose (RfD), the daily dose expected to be without harm. PFOA and PFOS assessments utilized various UFs to account for: potential differences in sensitivity among people (intraspecies UF) and between humans and animals (interspecies UF); gaps in toxicity data (database UF); and critical effect studies for which the POD was a LOAEL (LOAEL-to-NOAEL UF). UFs were applied differently across PFOA and PFOS assessments. The EPA and all state-based PFOA assessments except for North Carolina have total UFs of 300. North Carolina, the state with the highest proposed PFOA guideline level, has a total UF of only 30 based on intraspecies and interspecies UFs. For PFOS, Texas and Minnesota have total UFs of 100 while other states and the EPA have total UFs of 30. Texas includes a UF for LOAEL-to-NOAEL extrapolation, and Minnesota a database UF to account for potentially more sensitive immune effects.

States and EPA developed guideline levels that are based on a single critical effect but are intended to also be protective of other cancer and non-cancer health outcomes. Though New Jersey's recommended PFOA MCL is based on an RfD for liver toxicity, the state also considered whether the MCL would be protective for cancer endpoints or mammary gland development. Their assessment based on increased incidence of testicular tumors in rats arrived at the same 14 ng/L guideline level [45]. Their assessment based on altered mammary gland development produced a recommended PFOA MCL equivalent to 0.77 ng/L—18 times lower than the RfD used to derive the proposed MCL. This lower MCL was not recommended due to the lack of precedent for mammary gland development as a critical endpoint in risk assessment, although an additional UF of 10 for sensitive effects was applied to protect for this endpoint [45]. Vermont and EPA both calculated PFOA guideline levels for testicular cancer and determined that guideline levels based on the non-cancer endpoints were more protective. Minnesota did not derive a cancer-based PFOA guideline level, instead concluding that existing data were inadequate for assessing carcinogenic potential and that the non-cancer guideline was protective of potential cancer effects. All PFOS guideline levels are based on non-cancer endpoints, with most assessments indicating that cancer endpoints were reviewed and found to be not sufficiently well-studied to establish a cancer-based guideline level.

Exposure assessment

Following the derivation of an RfD, exposure assumptions are used to establish a concentration in drinking water

that is intended to be health protective, usually targeted to protect sensitive subgroups such as children. Exposure assessment relies on assumptions about the target population, water ingestion rates, and proportion of the daily dose supplied by drinking water relative to other exposure sources, known as the relative source contribution (RSC). These assumptions may vary based on the type of guideline (e.g., groundwater or drinking water).

In PFOA and PFOS assessments, target populations to be protected differed across states, even among those that used the same critical endpoint and/or had a similar RfD. EPA, Alaska, and Vermont derived the same critical endpoint and RfD for PFOA, yet their guideline levels ranged from 20 ng/L (Vermont) to 400 ng/L (Alaska), a 20-fold difference, because they used different exposure parameters. Vermont and EPA selected different target populations (infants for Vermont, lactating women for EPA), leading to divergent water ingestion rates and consequently different PFOA guideline levels for water. Minnesota's assessment is based on exposure for breastfed and formula-fed infants. Texas assumed that children's water consumption is 0.64 L/day, while Alaska assumed it is 0.78 L/day.

States also differed in their selection of RSC values. Most states and EPA assumed an RSC value of 20% for drinking water, which limits daily exposure from contaminated drinking water to 20% of the RfD so that additional exposures from other sources, such as consumer products or diet, do not push total exposure above the RfD. All other exposure assumptions being equal, lower RSC values correspond to lower drinking water guideline levels. Minnesota and Maine used human biomonitoring studies to derive RSCs for PFOA and PFOS ranging from 20% to 60%. Alaska and Texas used a 100% RSC, meaning that for people drinking water at their guideline, any dietary and consumer product exposures would raise their intake above the RfD. The Alaska and Texas PFOA and PFOS guidelines, which are 4–8 times higher than EPA's HAs, were developed for remediation and clean-up of contaminated sites, and these states use EPA's HAs as limits for PWS drinking water.

Factors contributing to variation in PFAS guideline levels

Considering the most recent adopted or proposed PFOA and PFOS water guideline levels at the federal and state levels, the range of "safe" levels in drinking water spans almost two full orders of magnitude, from 13 to 1000 ng/L. This variation reflects responses to scientific uncertainty in risk assessment, technical decisions and capacity, and social, political, and economic influences from involved stakeholders.

Scientific decisions

Differences between water guidelines in part reflect responses to scientific uncertainty. As described above, health risk assessment requires many assumptions and estimates in order to predict a safe exposure for humans. These include identifying critical effects, addressing interspecies and intra-species variation, quantifying other uncertainties, and selecting exposure parameters. Many areas of toxicity and exposure research on PFAS have not achieved scientific consensus so risk assessors make diverse choices.

Another important consideration in these and future assessments is the consideration of epidemiological evidence. Many of the assessments noted that effects in human studies were consistent with the critical effect in animal studies, giving greater confidence to the assessment. However, all of the assessments used dose-response data from animal studies as a basis for their drinking water levels. New Jersey assessments compared their target PFOS serum level of 23 ng/mL with the midrange of serum levels in epidemiological studies that reported effects (6–27 ng/mL) and with U.S. serum levels (median 5 ng/mL, 95%ile 19 ng/mL, from 2013–2014 NHANES) [46]. Based on this comparison, New Jersey recognized the need to minimize any additional exposures from drinking water since the population is already approaching effect levels from the epidemiological studies and risk-based exposure limits. While risk assessors generally expect their approaches to produce exposure levels that will be protective for exposed humans, PFOS immune effects in children are reported at lower exposures than the EPA's drinking water advisory levels [46]. A recent assessment used epidemiological data to propose a drinking water guideline of 1 ng/L to prevent additional increases in serum PFOS levels [47]. Several other endocrine disrupting compounds show effects in humans at exposures below EPA risk-based exposure limits, including di-(2-ethylhexyl) phthalate (DEHP) and polybrominated diphenyl ethers (PBDEs) [48].

The number of peer-reviewed scientific articles on PFAS has increased dramatically since 2000, while federal and state PFAS drinking water guideline levels have generally decreased over this time (Fig. 1). This demonstrates a common phenomenon: initial risk assessments based on limited data are often shown not to be health protective once more complete data become available. For PFOA and PFOS, the tightening of the guidelines is largely not due to new toxicology studies, but rather to improved exposure research, advances in analytical measurement technologies, improved biomonitoring and toxicokinetic data, and epidemiological findings. For example, both of EPA's PFOA HAs, the 2009 provisional HA for short term exposure and the 2016 lifetime HA for chronic exposure, are based on

developmental effects from the same mouse study [49], but different exposure parameters and toxicokinetic assumptions led to a much lower HA in 2016. Seven of the eight PFOA assessments, all released between 2012 and 2017, use critical endpoints from studies published in 2006 or earlier. EPA's assessments are also influential: once EPA derived RfDs for the 2016 HAs, states such as Minnesota and Vermont used these RfDs along with different decisions about exposure parameters, resulting in lower guideline levels.

The most sensitive toxicological endpoints—altered mammary gland development and suppressed immune function—were not the basis for EPA's PFOA and PFOS HAs. However two states, Texas and New Jersey, did use these endpoints as the basis for their PFOA Protective Concentration Level (PCL) and PFOS MCL, respectively. Although in utero PFOA exposure has been shown to alter mammary gland development in rodents [50, 51], this specialized endpoint is not routinely evaluated in regulatory toxicity studies and there is limited precedent for using it in risk assessment [52, 53]. To the best of our knowledge, altered mammary gland development has never been used as a critical endpoint for the basis of any federal regulatory risk assessment in the United States.

Texas based their PFOA PCL on altered mammary gland development from a full gestational study in mice since this endpoint showed a dose response. Texas determined this RfD to be protective of increased liver weight effects observed in several other studies. New Jersey's PFOA assessment did not use mammary gland changes as the critical effect but did recognize that it was most sensitive and included an additional UF for database uncertainty related to mammary gland effects. Minnesota identified delayed mammary gland development as a co-critical effect, but did not include additional UFs. North Carolina and EPA cited uncertainty related to variation in response between mouse strains, inconsistent methods across studies, and questions about toxicokinetics as challenges for using this endpoint [35, 54], though risk assessments commonly rely on endpoints for which there is substantial intra- and inter-species variation in sensitivity. Most notably, EPA discounted effects on mammary gland development because these alterations were not associated with decreased lactation function and the mode of action for mammary gland development effects is not well described. Though EPA was reluctant to consider the changes adverse, a substantial body of scientific work suggests that altered mammary gland development is likely to influence later breast cancer risk [53]. New research to better characterize these associations is important because many endocrine disruptors alter mammary gland development if exposure occurs in utero or early in life. Routine assessment of mammary gland development in toxicity studies of endocrine disruptors will be informative and improve understanding of these changes and reduce uncertainty for future risk assessments.

New Jersey used decreased plaque forming cell response (suppressed immune function) as the basis for their PFOS MCL, noting also the consistency between this effect and decreased vaccination response in epidemiological studies. Minnesota identified suppressed immune function as a co-critical effect and included a database UF of 3 for immunotoxicity. While the EPA indicated a concern for adverse immune effects, it chose not to use suppressed immune function as the basis for the PFOS HA because a "lack of human dosing information and lack of low-dose confirmation of effects in animals for the short-duration study precludes the use of these immunotoxicity data in setting the RfD" [35]. The New Jersey assessment includes a rebuttal of EPA's decision, noting that EPA has used this endpoint as a basis for RfDs for other chemicals [46].

Social, political, and economic influences

While risk assessments such as these PFAS water guidelines are presented as being based solely on scientific considerations, this process is also influenced by political, social, and economic factors [55–59]. For PFAS, much like other high-value products such as tobacco, the landscape of what is scientifically known and unknown about their health and environmental impacts is influenced by the context of knowledge production. Internal industry documents reveal a broad "science-based defense strategy" to "command the science" on PFAS, ranging from suspected influence on state environmental protection agencies in the case of West Virginia, to the selective peer review publication of internal research, to paying academic scientists to influence the peer-review process [10, 60, 61].

PFAS manufacturing companies have influenced PFAS water guidelines in both overt and subtle ways. For example, in 2001 EPA and West Virginia Department of Environmental Protection (WVDEP) learned that DuPont scientists had found high levels of PFOA in regional drinking water. The following year, DuPont collaborated with WVDEP and a state-appointed C8 Assessment Toxicity Team to develop a screening level of 150,000 ng/L, despite numerous conflicts of interest and DuPont's own internal guideline of 1000 ng/L [10, 62].

Economically invested corporations have indirectly influenced the development of PFAS drinking water guideline levels through the strategic production and dissemination of industry-friendly research, a well-documented pattern in environmental health [63]. Recent litigation by the State of Minnesota Attorney General against 3M revealed internal correspondence between the company and academic scientists paid as consultants. In one

instance, an academic scientist hired by 3M wrote in private emails that he intentionally described his work reviewing articles for publication as "literature reviews" in order to avoid a paper trail to 3M, bragged about rejecting an article on PFAS health effects, and offered to pass unpublished articles to peer reviewers recommended by 3M, clear violations of scientific norms [60].

Industry sponsorship of toxicological research and risk assessments can also influence the developments of guidelines through the "funding effect" in which funding source influences published outcomes [64–66]. Studies or assessments funded by a company or industry that benefits financially from the product under investigation are less likely to identify risks and more likely to demonstrate efficacy (or ambiguity), while the opposite is true of studies funded by government agencies or independent parties. Of the eight critical studies used to derive PFOA ($n = 5$) or PFOS ($n = 3$) guidelines, five were conducted by PFAS manufacturers (3M or DuPont), two were conducted by the U.S. government (EPA or NIEHS), and one was conducted by academic researchers with funding from the Chinese government. North Carolina's PFOA guideline, the highest in the country, heavily references a risk assessment conducted by industry consultants [67]. However, the small number of PFAS guidelines prevents any quantitative analysis of funding effects. Risk assessments, which rely on many assumptions to estimate human exposure and toxicity in the absence of data, are more vulnerable to funding effects. For example, a 2009 PFOA risk assessment funded by DuPont and 3M identified 880 ng/L as "a reliable, albeit conservative" level for an MCL, over 12 times higher than the EPA HA [67].

Industry-funded research may also influence the overall landscape of PFAS research because it is selectively produced and shared [10]. For example, most research conducted by chemical companies is never published or made public, even when disclosure could be useful for assessing chemical risk. Major PFAS manufacturers have repeatedly violated information disclosure requirements under the Toxic Substances Control Act (TSCA) Section 8(e) by not disclosing information on substantial risks related to PFAS in production [68, 69]. This practice has resulted in multimillion dollar fines and also delayed the production of science on environmental and human health effects of PFAS by decades [70, 71]. Today, PFAS manufacturers commonly assert that information on production quantities, use in consumer goods, and chemical identity is confidential business information, creating barriers for scientists and regulators seeking to prevent harmful exposures.

Unlike some states where limited regulatory appetite and strong industry and political influence may slow progress on protecting public health by establishing drinking water exposure limits, other states have developed more protective and scientifically sound PFAS guideline levels in response to significant public and community pressure. After communities in Vermont learned of water contamination, social pressure led to state guidelines for PFOA and PFOS that were lower than EPA's [72]. In contrast, North Carolina, home to a major Chemours PFAS manufacturing facility, has not updated their PFOA interim maximum allowable concentration of 2000 ng/L, the highest in the United States, despite a 2012 proposal that this guideline be lowered to 1000 ng/L. North Carolina recently developed the nation's first drinking water provisional health goal for GenX (hexafluoropropylene oxide dimer acid), a PFOA replacement, following discovery of widespread contamination in local rivers that are used for drinking water [73]. This example demonstrates that local pollution concerns can motivate states to develop guidelines or standards without waiting for federal precedent. Legislators at the state and federal level may play an increasing role going forward. Recent examples include a legislatively proposed 5 ng/L level for PFOA and PFOS in Michigan and pressure from 25 U.S. Senators on EPA to develop a PFAS MCL [33, 74].

Discussion and conclusion

The wide range of PFOA and PFOS guidelines—up to 70-fold difference between states—as well as the lack of enforceable MCLs and deference by many states to EPA's HA of 70 ng/L have significant public health implications. Our finding that some states have taken additional steps beyond federal action in evaluating and/or regulating PFAS is consistent with states taking more health-protective action on other chemicals, including flame retardants and bisphenol A [75, 76].

EPA's HAs do not require ongoing monitoring by PWSs or treatment of water that exceeds the HAs, though in practice many other entities use the HA to make remediation decisions. If MCLs existed for PFAS, regulators would have greater authority to take action at contaminated sites under CERCLA, and DoD sites would be able to move forward with remediation of contaminated sites [33]. In addition, given the toxicity, persistence, and mobility of PFAS, systematic screening of PWSs is a logical approach to protect public health. Some states, including Michigan and Washington, are testing PWSs for certain PFAS [77, 78], and New Jersey's recommended MCLs would require routine testing. In the absence of MCLs, guidelines are applied only after contamination is discovered by other mechanisms, for example, when residents seek water testing near known industrial sites. Public and regulatory awareness of PFAS water contamination has benefited from nationwide testing initiatives, including EPA's UCMR

testing and DoD identification of PFAS-contaminated military sites. The recently authorized nationwide study on PFAS exposure at military sites may be particularly useful in raising awareness and potentially supporting further regulatory action [79].

Regulatory and scientific attention to PFAS has focused on PFOA and PFOS, but the scope of potential PFAS contamination is much broader. While there are data available to support risk assessment for several additional PFAS, including perfluorobutyrate (PFBA), perfluorobutanesulfonic acid (PFBS), perfluorononanoic acid (PFNA), perfluorohexane sulfonic acid (PFHxS), and GenX, there are no studies on prevalence, exposure, and toxicity for many other PFAS, or even analytical methods to detect them [22]. PFAS as a class are generally persistent and mobile, and the few that have been adequately tested share some toxic effects and exposure characteristics with PFOA and PFOS [14, 18–21, 80]. The lack of information and potential scope of the contamination poses significant challenges for protecting public health. The fact that several guideline levels, including EPA's HAs, apply to the total concentration of multiple PFAS suggests that regulatory agencies are attentive to PFAS as a class, not just as individual compounds. In the absence of toxicity data on individual chemicals, regulators could use well-characterized PFAS as analogues for deriving RfDs and guideline levels, or could develop methods to regulate PFAS as a class, although this would involve additional assumptions and uncertainties. Texas developed PCLs for 16 PFAS, deriving RfD values for PFAS with limited toxicity data using well-characterized PFAS as surrogates [81]. Relative potency estimates have been used in other chemical classes, such as polycyclic aromatic hydrocarbons and dioxins, and are being explored for PFAS [82]. Some existing regulations treat all long-chain PFAS similarly. The U.S. Food and Drug Administration (FDA) has restricted all long-chain PFAS as a class [55, 83], and EPA's PFOA Stewardship Program includes PFOA and all "precursor chemicals that can break down to PFOA, and related higher homologue chemicals" [84]. The similarities between many PFAS in terms of chemical structure and exposure potential, combined with potential differences in toxicity and the long time required to gather sufficient data, further raise the importance of limiting manufacture and use of PFAS before they become exposure concerns.

EPA-validated drinking water testing protocols exist for 18 PFAS (EPA Method 537), though validated methods are lacking for other PFAS and other media, such as groundwater. It is difficult to understand why EPA has not included any PFAS in the fourth cycle of UCMR testing, despite significant data gaps regarding the extent of drinking water contamination with other PFAS and the need for surveillance using lower detection limits [85]. The focus of current water screening and treatment efforts solely on removing PFOA and PFOS is concerning because carbon filtration designed to remove long-chain PFAS is less effective at removing short-chain PFAS and PFAS transformation products likely present in AFFF-contaminated water [86] and at PFAS production sites [21].

Our review of PFAS drinking water guideline levels highlights opportunities to extend risk assessment methods to include some important endpoints such as mammary gland development and immune function. Reports of immunosuppression in children with exposures within the exposure range prevalent in the general population have raised concern that EPA's HAs are not adequately protective, since modeling indicates that consumption of drinking water at 70 ng/L would substantially increase PFOA and PFOS blood levels above current U.S. background levels [47]. Additionally, New Jersey's PFOA assessment estimated that the RfD for mammary gland changes is below median blood levels in the general population [45]. Grandjean and Clapp [47] proposed that a drinking water concentration of 1 ng/L for PFOA and PFOS would not be expected to lead to an increase in population-level blood serum levels above current U.S. averages.

Our analysis also highlights opportunities to consider epidemiological data more carefully in conjunction with toxicological and exposure data. Despite a relatively robust epidemiological literature for PFOA and PFOS, only New Jersey showed how their target blood level was in the range of exposures in human studies that show effect on vaccine response. New Jersey also used human biomonitoring data to illustrate that even small increases in exposure are problematic because current exposure levels are near levels associated with health effects [22]. However, the environmental co-occurrence of multiple PFAS is a challenge for using epidemiological data to develop guideline levels for individual PFAS [87]. Considering information from human biomonitoring and epidemiology adds important context to the risk assessment process.

The scientific and regulatory landscape on PFAS continues to evolve rapidly. Advances in analytical methods and decreased cost of measuring certain PFAS in water and other media broaden the ability of PWSs, regulatory and health agencies, academics, and nonprofits to identify water contamination. In June 2018, the Agency for Toxic Substances and Disease Registry (ATSDR) released a draft Toxicological Profile that derived minimal risk levels (MRLs), which are similar to RfDs, for intermediate duration exposure (15–364 days) of four PFAS routinely measured in NHANES [28]. The MRL values for PFOA (3 ng/kg/day) and PFOS (2 ng/kg/day) are 6.7 and 10 times lower than the RfDs EPA used to develop its 2016 HAs and similar to those developed by New Jersey, though they are

based on different studies and endpoints. The release of this report became surrounded in controversy amidst suggestions that months earlier, EPA and other government officials sought to delay its release, citing concerns about public reaction [88], and demonstrates how political and economic factors can affect the timely development of health-protective guidelines.

In the absence of enforceable, nationwide water standards for PFAS, some states have developed more health-protective and scientifically sound guidelines. This may create or exacerbate public health disparities because not all states have the resources to develop guideline levels. The ability of states to develop their own guideline levels and standards provides diverse risk assessment approaches as models for other state and federal regulators, while a sufficiently protective, scientifically sound, and enforceable federal standard would provide more consistent protection.

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Compliance with ethical standards

Conflict of interest The authors declare they have no conflict of interest.

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Perfluorinated Alkyl Substances: Emerging Insights Into Health Risks

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Abstract

Perfluorinated alkyl substances have been in use for over sixty years. These highly stable substances were at first thought to be virtually inert and of low toxicity. Toxicity information slowly emerged on perfluorooctanoic acid and perfluorooctane sulfonate. More than thirty years ago, early studies reported immunotoxicity and carcinogenicity effects. The substances were discovered in blood samples from exposed workers, then in the general population and in community water supplies near U.S. manufacturing plants. Only recently has research publication on perfluorooctanoic acid and perfluorooctane sulfonate intensified. While the toxicology database is still far from complete, carcinogenicity and immunotoxicity now appear to be relevant risks at prevalent exposure levels. Existing drinking water limits are based on less complete evidence that was available before 2008 and may be more than 100-fold too high. As risk evaluations assume that untested effects do not require regulatory attention, the greatly underestimated health risks from perfluorooctanoic acid and perfluorooctane sulfonate illustrate the public health implications of assuming the safety of incompletely tested industrial chemicals.

Keywords

carcinogen, exposure limit, immunotoxicant, perfluorinated octanoic acid, perfluorooctane sulfonate, risk assessment

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Introduction

Poly- and perfluorinated alkyl substances (PFASs) have been in use for over sixty years.[1] First manufactured by the 3M Company in Cottage Grove, Minnesota, perfluorooctanoic acid (PFOA) was a primary PFAS product, but perfluorooctane sulfonate (PFOS) and other PFASs were also produced. By about 2000, their global environmental dispersion became publicly known. A phase-out of commercial PFOS production by the end of 2002 was announced by 3M in 2000, and eight major U.S. producers have agreed to phase out PFOA no later than 2015. Recent reports on adverse effects[2,3] suggest that the toxicity of these substances has long been underestimated.

The PFAS show high thermal, chemical, and biological inertness—properties that make them useful for certain industrial purposes, but persistence may also create an environmental hazard.[4] The strong carbon–fluorine bond renders the PFASs highly persistent in the environment and in the human body. However, the functional group at the end of the perfluorinated carbon chain made the PFASs far from inert. By the 1970s, the physical and chemical properties were well known.[5,6] Thus, many PFASs can leach through soil to reach the groundwater, while some PFASs may evaporate and disseminate via the atmosphere.[7] Although most of them are oleophobic and do not accumulate in fatty tissues (unlike dioxins and other persistent halogenated compounds), they were later found to bioaccumulate in aquatic and marine food chains, especially PFOS.[8] Thus, as criteria for persistent, bioaccumulative, and toxic chemicals were developed and refined in the 1990s,[9] the PFAS physical and chemical properties should have raised warning signs.

Little was published in scientific journals on PFAS toxicology until the 1980s perhaps because compounds resistant to breakdown were erroneously considered inert.[10] The present overview relies on recent reviews, such as the Agency for Toxic Substances and Disease Registry (ATSDR) draft toxicological profile,[7] a draft risk assessment developed by the U.S. Environmental Protection Agency (EPA), and recent overviews.[2,11–13] Our objective is to illustrate the problems that can result from the regulatory assumption that untested chemicals are safe. We focus on PFOS and PFOA as the substances with the best available information to review the emergence of new insight into carcinogenicity and immunotoxicity as potential critical effects.[2,14] We focus our comments on these two effects because of their long history of scientific study, while recognizing that other adverse health effects have recently been documented.[70] Although mainly relying on published information, we are aware that a major chemical company was fined by the U.S. EPA for failing to comply with the legal requirement of reporting information to the EPA about substantial risk of injury to human health or the environment due to PFAS.[15] A chronology of important events in understanding PFAS's health risks is provided in Table 1.[16]

Table 1. Time Course of Important Developments Regarding PFAS Exposure and Health Risks.

Year	Event
1947	PFAS production starts at 3 M plant in Cottage Grove, Minnesota
1962	Internal DuPont document raises concern about health risks
1970s	PFAS vapor pressures and water solubilities in chemical handbooks
1978	Unpublished monkey study reveals immunotoxicity and other adverse effects due to PFOA
1980	Organic fluoride determined in serum from production workers
1981	Concern about birth defects in children of female production workers
1987	PFOA carcinogenicity reported in rat study
1993	3 M begins to monitor PFOA in serum from production workers
	Mortality study shows excess occurrence of prostate cancer
1998	Serum from U.S. blood donors shown to contain PFAS
2000	Global dissemination of environmental PFAS contamination documented
	3 M announces plan to phase out commercial production of PFOS
2005	Extensive drinking water contamination discovered in Minnesota
2008	Health risk limits for PFAS in drinking water are issued
	Mouse study shows immunotoxicity at serum PFAS concentrations similar to human exposures
2010	Decrease of PFOA emissions by 95% said to be completed
2011	PFOA induces delayed mammary gland development in mice at low exposures
2012	PFAS immunotoxicity reported in children

Note. Adapted from Grandjean and Clapp.[16]
PFOA = Poly- and perfluorinated alkyl substances; PFOS = perfluorooctane sulfonate.

Human Exposure to Perfluorinated Compounds

The existence of PFASs in the human body was first suspected in the late 1960s when fluoride in blood samples was found to be partially bound to organic compounds of unknown structure.[17] High concentrations in exposed workers were documented in the 1970s,[18] and specific PFASs were later identified in serum samples from workers at production facilities[19] in accordance with the ready absorption of the compounds in laboratory animals after oral or inhalation exposure.[20]

Multiple sources play a role for exposures of the general population, and human exposures include precursor compounds that may be broken down into PFOA and PFOS.[1] In the Mid-Ohio Valley of the United States, drinking water supplies were contaminated with PFOA in the 1980s from an industrial facility,[21] and aquifers in Minnesota were also contaminated from a production

plant.[22] Concentrations of PFOA in many water samples exceeded 1 μg/L (1000 ng/L), with concentrations of PFOS being almost as high.[7] Other routes of human exposure are primarily from consumer product use, and degradation or improper disposal of PFAS-containing materials, including food wrapping.[1,23,24]

Analysis of serum samples from the National Health and Nutrition Examination Survey (NHANES) about year 2000 showed that PFOS and PFOA were detectable in all Americans.[25] Median concentrations in serum were about 30 ng/mL (PFOS) and 5 ng/mL (PFOA). The average had decreased eight to ten years later to less than half for PFOS, while PFOA had changed much less.[26,27] PFASs are transferred through the human placenta and via human milk.[28,29] Overall, serum concentrations in children tend to be higher than in adults.[30]

Serial analyses of serum samples from former 3M production workers after retirement suggested elimination half-lives for long-chain PFASs to be ~three years (PFOA) and ~five years (PFOS).[31] Declines in serum-PFOA concentrations after elimination of the water contamination suggest a median elimination half-life of 2.3 years,[32] thus confirming the persistence of PFAS in the human body.

Adverse Health Effects

The main evidence on adverse effects in humans comes from observational studies of cohorts of production workers and community studies of subjects exposed either at background levels or through contaminated drinking water. Some studies are hampered by imprecise estimates of long-term PFAS exposures and may for this reason have underestimated the effects.[33] Follow-up studies of workers have largely shown an overall mortality deficit,[34–36] thus most likely reflecting the presence of a "healthy worker" effect.[37]

New evidence has emerged, as a settlement agreement in 2005 established the C8 Health project, where data on approximately 70,000 exposed Ohio and West Virginia residents provided information on drinking water intake, measured and calculated serum-PFOA concentrations, and a variety of possible clinical outcomes.[38,39] Additional evidence on associations between PFAS exposure and disease parameters in the general population comes from the National Health and Nutrition Examination survey database, which provides national data for exposures to environmental chemicals that can be linked to concurrent health information on the study participants.[25]

With regard to experimental toxicity studies, most published reports are based on the rat, which eliminates PFAS much more rapidly than humans and, therefore, is not an ideal species.[12] Even today, chronic toxicity studies in other species are lacking, and a formal cancer bioassay has not yet been completed. In addition, insufficient attention had been paid to exposures during sensitive developmental stages.

Cancer. The rodent cancer bioassay has long served as a key component of carcinogenicity assessment.[40] Evidence on cancer risks in rodents exposed to PFASs and other peroxisome proliferating substances, which promote rapid cell division, originates from the late 1970s, specifically in regard to pancreatic tumors and hepatocellular carcinomas.[41–43] For Leydig cell tumors, the first evidence describing the tumor mechanisms was published in 1992,[44] and further review of cancer mechanisms appeared in the late 1990s.[45]

The DuPont cancer surveillance system has been monitoring cancer incidence in workers as far back as 1956,[46] and an internal report showed increased leukemia incidence in employees at a PFOA production plant. As a result of the 3M findings and animal carcinogenicity studies showing increased male reproductive organ cancer, prostate cancer has been monitored in DuPont workers from 1998, although the results have apparently not been released. An updated cancer surveillance report covered the years 1956 to 2002 showed excess kidney cancer (Standardized Incidence Ratio [SIR] = 2.3, 95% confidence interval [*CI*]: 1.36–3.64), bladder cancer (SIR = 1.93, 95% *CI*: 1.14–3.06), and myeloid leukemia (SIR = 2.25, 95% *CI*: 1.03–4.28) in the employees, and an elevated, but not statistically significant, risk of testicular cancer (SIR = 1.46, 95% *CI*: 0.47–3.41).[47]

Initially, the most important 3M worker study was Frank Gilliland's thesis project on retrospective mortality of 2788 male and 749 female production workers during 1947 to 1984. Based on four cases, an excess occurrence of prostate cancer was found (Standardized Mortality Ratio [SMR] = 3.3, 95% *CI*: 1.02–10.6) in PFOA-exposed workers with greater than ten years of employment.[34] There were subsequent analyses of cancer in 3M workers after reported further evidence of increased prostate cancer risk but not for other cancers.[48,49] The key epidemiologic studies are summarized in Table 2. Incomplete follow-up, uncertainties in exposure assessment, and incomplete ascertainment of cancer mortality limit the conclusions that can be drawn from this evidence.

The EPA draft risk assessment of PFOA reviewed the published animal and human epidemiologic studies up to 2005 and concluded that the evidence was "suggestive" of a cancer risk in humans. When reviewing the same evidence a year later, the majority of an expert committee recommended that PFOA be considered "likely to be carcinogenic to humans".[50]

This conclusion is supported by the recent C8 Health Project results.[51] Thus, two different epidemiological approaches[52,53] support the association between PFOA exposure and both kidney and testicular cancer and suggest associations with prostate and ovarian cancer and non-Hodgkin lymphoma. The C8 Science Panel specifically listed kidney cancer and testicular cancer as having a "probable link" to C8. Although PFOA should therefore be considered a "likely" human carcinogen based on sufficient evidence in experimental animals and limited evidence in human epidemiology studies, current regulations of PFASs are based not on carcinogenicity but on developmental toxicity and changes in liver weight.

Table 2. Summary of Main Cancer Epidemiology Studies.

Reference	Study population	Main results	Comments
[34]	2788 male and 749 female workers in PFOA production plant	Male all cause SMR = 0.77 (95% *CI*: 0.69–0.86); Prostate cancer SMR = 3.3 (95% *CI*: 1.02–10.6) with 10 + years employment	Likely healthy worker effect; six prostate cancer deaths overall
[48]	2083 production workers employed at least one year in Alabama PFOS fluoride production plant	All cause SMR = 0.63 (95% *CI*: 0.53–0.74); bladder cancer SMR = 16.12 (95% *CI*: 3.32–47.14) in those with high exposure jobs	Likely healthy worker effect; small number of cancer deaths, only three bladder cancer deaths
[35]	6027 workers who worked in DuPont West Virginia plant between 1948 and 2002	All cause SMR = 67 (95% *CI*: 62–72); all cancer SMR = 74 (95% *CI*: 65–84); kidney SMR = 152 (95% *CI*: 78–265)	Likely healthy worker effect; comparison to other DuPont Region 1 workers unremarkable
[49]	3993 workers employed at least a year in Minnesota PFOA plant between 1947 and 1997	All cause SMR = 0.9 (95% *CI*: 0.7–1.1); prostate cancer SMR = 2.1 (95% *CI*: 0.4–6.1); moderate/high exposed SMR = 3.2 (95% *CI*: 1.0–10.3)	Suggestive increased mortality from bladder cancer and cerebrovascular disease
[51]	5791 workers exposed to PFOA in DuPont West Virginia plant	All cause SMR = 0.98 (95% *CI:* 0.92–1.04); Kidney cancer SMR = 2.66 (95% *CI*: 1.15–5.24) in most highly exposed quartile	Detailed exposure estimates, additional results with lagged analyses for mesothelioma and chronic renal disease deaths
[52]	Cancer cases and controls from five West Virginia and Ohio counties diagnosed 1996–2005	Kidney cancer *OR* = 2.0 (95% *CI*: 1.0–3.9) for very high exposure category; testis cancer *OR* = 2.8 (95% *CI*: 0.8–9.2) for very high exposure category	Community water contamination estimates showed suggestive associations with several types of cancer

Note. PFOA = Poly- and perfluorinated alkyl substances; PFOS = perfluorooctane sulfonate.

Mechanisms of cancer development are now being explored.[2,54] Among possible mechanisms, induction of hormone-dependent cancer has been suggested in rodent studies.[55] Developmental exposure to PFOA induces effects that are not necessarily seen in response to exposures during adulthood,[55] as reflected by endocrine disruption effects in humans exposed to PFASs during early development.[56,57]

Immunotoxicity. Among early toxicology studies,[20] immunotoxicity was considered as a main effect in a rhesus monkey study sponsored by 3M,[58] although the report was not published in the open literature. Four monkeys exposed to subacute toxicity from the ammonium PFOA salt showed atrophied thymus, diffuse atrophy of lymphoid follicles of the spleen, and other signs of immunotoxicity. Researchers at the time were well aware of the adverse effects to the "reticuloendothelial system," and increasing attention was being paid to adverse effects on immune functions.[59] However, these findings did not lead to further exploration of immunotoxic risks associated with PFAS exposure until decades later. Routine parameters, such as spleen microscopy and general clinical chemistry, failed to show any significant effects in non-human primates.[60]

In recent years, immunotoxicity of poly- and perfluorinated compounds has been demonstrated in a wide variety of species and models.[14] In the mouse, PFOA exposure caused decreased spleen and thymus weights, decreased thymocyte and splenocyte counts, decreased immunoglobulin response, and changes in specific populations of lymphocytes in the spleen and thymus.[7,14] Reduced survival after influenza infection was reported in mice as an apparent effect of PFOS exposure.[61] When injection of sheep erythrocytes was used as antigen exposure in the mouse model, the lowest observed effect level for a deficient antibody response corresponded to average serum concentrations of 92 ng/g and 666 ng/g for male and female mice, respectively.[62] These serum concentrations are similar to or slightly exceed those prevalent in residents exposed to contaminated drinking water.[21,63,64] Although a 3M-supported study reported no immunological effects at a high dietary PFOS exposure in the same strain of mice,[65] another study of gestational exposure confirmed that male pups were more sensitive than females and that developmental exposure can result in functional deficits in innate and humoral immunity detectable at adulthood.[66]

In human studies, childhood vaccination responses can be applied as feasible and clinically relevant outcomes because children have received the same antigen doses at the same ages.[67] In the fishing community of the Faroe Islands, PFOS in maternal pregnancy serum showed a strong negative correlations with antibody concentrations in 587 children at age of five, where a doubling in exposure was associated with a difference of −41% ($p = .0003$) in the diphtheria antibody concentration.[3] Poly- and perfluorinated compounds in the child's serum at age five showed negative associations with antibody levels at age seven, and a doubling in PFOS and PFOA concentrations was associated with differences in antibody

levels between −24% and −36% (joint effect of −49%, $p = .001$). For doubled concentrations at age five, PFOS and PFOA showed odds ratios between 2.4 and 4.2 for falling below a clinically protective antibody level of 0.1 IU/mL for tetanus and diphtheria at age seven.[3] Serum concentrations of both PFASs are similar to, or lower than, those reported from the U.S. population.

A study of ninety-nine Norwegian children at age three found that maternal serum PFOA concentrations were associated with decreased vaccine responses, especially toward rubella vaccine, and increased frequencies of common cold and gastroenteritis.[68] In a larger study, PFOS and PFOA concentrations in serum from 1400 pregnant women from the Danish National Birth Cohort were not associated with the hospitalization rate for infectious disease (including such diagnoses as pneumonia or appendicitis) in 363 of the children up to an average age of eight.[69] In adults, PFOA exposure was associated with lower serum concentrations of total IgA, IgE (females only), though not total IgG.[70] In the exposed Ohio Valley population, elevated serum-PFOA concentrations were associated with reduced antibody titer rise after influenza vaccination.[71] Taking into account the likely sensitivity of the various outcome measures as indication of PFAS immunotoxicity, the combined human and experimental evidence is in strong support of adverse effects on immune functions at current exposure levels.

With regard to mechanisms of immunotoxicity, PPAR receptor activation may play a role.[7,14] However, experimental evidence suggests independence of PPARα for at least some of PFOA's immunotoxic effects, as shown in PPARα knockout models.[72] White blood cells from human volunteers showed effects even at the lowest in vitro PFOS concentration applied, that is, 0.1 μg/mL (or 100 ng/mL).[73] This level is similar to concentrations seen both in affected male mice[62]and in U.S. residents exposed to contaminated drinking water.[21,63,64]

Implications for Prevention

The U.S. EPA first issued a draft risk assessment of PFOA in 2005, but a final, quotable version has yet to appear. While a reference dose is not available, the EPA in 2009 published provisional drinking water health advisories of 0.4 μg/L (400 ng/L) for PFOA and 0.2 μg/L (200 ng/L) for PFOS.[4] EPA used calculations of benchmark dose level (BMDL) from experimental toxicology studies and concluded at the time that "[e]pidemiological studies of exposure to PFOA and adverse health outcomes in humans are inconclusive at present." The same toxicology data published by the end of the last decade were used for derivation of drinking water limits authorized by United States and European union countries as well as the European union Tolerable Daily Intakes for PFOA and PFOS,[74] although different default assumptions and uncertainty factors were applied.

BMDL is recommended by the EPA and other regulatory agencies as a basis for calculations of safe levels of exposures.[75,76] As the BMDL is not a threshold, this lower 95% confidence limit is applied as a point of departure, and the guidelines proscribe a default tenfold uncertainty factor to be used for calculation of an exposure limit.

Table 3 lists relevant BMDL results in terms of serum concentrations. A sensitive outcome at first appeared to be the increase in liver weight; Leydig cell tumor formation was considered as a dose-dependent outcome and appeared to be less sensitive.[77] The same was true for immune system toxicity that was generally evaluated by differential leukocyte counts and microscopic examination of lymphoid tissues, sometimes complemented with a cell proliferation test;[78] functional tests were not conducted. In terms of serum concentrations, the BMDLs were 23 μg/mL serum for PFOA and 35 μg/mL for PFOS.[22] Expression of the BMDL in terms of the serum concentration is particularly useful, as it facilitates interspecies comparisons by taking into account toxicokinetic differences.

Recent data on mammary gland development in mice suggest that clear effects may result from much lower developmental exposures.[2] Benchmark dose calculations using a variety of models correspond to a serum concentration of 23 to 25 ng/mL,[12] that is, one-thousandth of the BMDL based on liver toxicity. Benchmark calculations are not available with regard to immunotoxic

Table 3. Benchmark Dose Level (BMDL) Results in Terms of Serum Concentrations of PFOA and PFOS.

Reference	Study type	BMDL	Outcome parameter
PFOA			
[77]	Adult rats with subchronic exposure	23,000 ng/mL	10% increase in liver weight
[2, 12]	Developmental exposure in mice	23–25 ng/mL	10% delay in mammary gland development
[3]	Prospective human birth cohort study	0.3 ng/mL	5% decrease in serum concentration of specific antibodies
PFOS			
[78, 85]	Adult cynomolgus monkeys with subchronic exposure	35,000 ng/mL	10% change in liver function and thyroid function
[3]	Prospective human birth cohort study	1.3 ng/mL	5% decrease in serum concentration of specific antibodies

Note. PFOA = Poly- and perfluorinated alkyl substances; BMDL = benchmark dose level; PFOS = perfluorooctane sulfonate.

effects in mice and cannot easily be estimated from published data[14] but would likely be orders of magnitude below previously calculated BMDLs.

Using the data from the recent study of immunotoxicity in children[3] and assuming a linear dose-dependence of the effects, BMDLs were calculated to be approximately 1.3 ng/mL for PFOS and 0.3 ng/mL for PFOA, both in terms of the serum concentration.[79] Using an uncertainty factor of ten to take into account individual susceptibility, the BMDLs would therefore result in a reference dose serum concentration of about or below 0.1 ng/mL. The experimental data require at least an additional interspecies threefold uncertainty factor for interspecies differences in toxicodynamics.[76] Thus, using a total uncertainty factor of 30, the reference dose based on mammary gland development in mice would correspond to a serum-PFOA concentration of 0.8 ng/mL. As the experimental studies that the regulatory agencies have relied upon so far correspond to serum concentrations 1000-fold higher, current limits for water concentrations of PFOS and PFOA appear to be too high by at least two orders of magnitude.

For comparison, an approximate limit for drinking water can be estimated by an independent calculation. PFOA concentrations in drinking water and in the serum of residents are highly correlated,[21,80] and the calculated ratio of 100-fold between the concentrations in the two media could therefore be used to calculate a concentration in drinking water that would correspond to the reference dose expressed in terms of the serum concentration. Assuming no other sources of exposure, a serum concentration of 0.1 ng/mL would correspond to a water concentration of approximately 1 ng/L, or 0.001 μg/L. Although neither of the two sets of calculations in any way represents a formal risk evaluation, it is noteworthy that current limits are generally several 100-fold higher than recent BMDL results would seem to justify.

Discussion

The PFASs have been in use for many decades, but their otherwise useful properties unfortunately result in persistence and dissemination in the environment. The toxic properties were initially explored in the 1970s, but the toxicological database has expanded only after environmental dissemination recently became known.

In the United States, the toxic substances control act Toxic Substances Control Act (TSCA) has been in force since the late 1970s but did not require testing of substances, such as PFASs, already in commerce at the time. Perhaps the TSCA even discouraged chemicals producers from testing substances that had already received blanket approval.[81] The voluntary decision in 2000 to phase-out PFOS production in the United States coincided with the first demonstration of environmental persistence and dissemination of PFASs.

Although comparatively few articles on PFASs were published in scientific journals prior to 2008,[82] our understanding of the toxicity of these compounds has its roots in studies already carried out in the late 1970s. Thus, more than thirty years ago, possible carcinogenicity and immunotoxicity had already been demonstrated in experimental studies, and they were complemented by internal company surveillance of birth defects, mortality, and clinical findings in workers. These reports could have inspired in-depth studies but apparently did not.

As judged from available publications, the early leads were not followed up with the focused research that in today's perspective would have seemed appropriate. Of note is also the EPA decision to fine a company for violation of the duty to report adverse effects of PFAS and the subsequent court-mandated health studies.[15,39] Had the first suspicions of health risks from PFAS exposures been explored in systematic research and testing, they could perhaps have triggered earlier and more vigorous efforts to control exposures to workers and to prevent community contamination and global dissemination.

The PFASs therefore provide an example of the "untested-chemical assumption" that the lack of documentation means that no regulatory action is required.[83] In this case, the assumption ignored preliminary evidence on plausible effects and did not inspire further exploration. The present overview suggests that these assumptions resulted in continued PFAS dissemination and exposure limits that may be more than 100-fold too high to adequately protect the general population against adverse health effects. Clearly, the absence of documentation from epidemiological studies should not be considered as a reason to conclude that adverse effects have not and will not occur.[84] Thus, the PFASs represent an example of a failed scientific and regulatory approach[83]and thereby also document the need for better linkage between research and risk assessment to inspire prudent chemicals control policies.

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Author Biographies

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From: Katie Weir
Sent: Tuesday, February 05, 2019 11:07 AM
To: Water <water@cityofmadison.com>
Subject: Contact the Water Utility Board

Hello,

I am writing to express my significant concern regarding the PFAS chemicals present in Well 15. I live in the Well 15 service area, and this issue is of particular concern to me because I have an 8 month old daughter. I have been breastfeeding my daughter since her birth, as is encouraged by health care professionals. However, this also means that the PFAS chemicals I have ingested through water provided by the city have been passed on to her. Infants are particularly susceptible to the negative health effects of toxins, including PFAS. It is heartbreaking to me to know that she is now at greater risk of several health conditions, including liver damage, thyroid disorder, and certain cancers, because I chose to breastfeed her - an act that is meant to provide health benefits to babies, not harm.

I understand that the EPA has set a limit that is well above the PFAS levels found in Well 15. There is still much research to do be done, however, about the actual "safe" level of PFAS in water, and many states have set lower limits. I do not understand why it is acceptable to have any amount of PFAS in our water.

My family is spending hundreds of dollars to have a reverse osmosis water filter installed in our home, in order to prevent further exposure to PFAS. We are fortunate that we are able to afford this. There are many families in the Well 15 area who may not be able to do the same thing to protect their children.

I ask that the Water Utility Board immediately take action to ensure there are no longer PFAS chemicals in our water.

Thank you,
Katie Weir
Madison, WI 53704

Statement on PFAS
Madison Water Utility Meeting, Feb 6, 2019

Perfluoroalkyl and polyfluoroalkyl substances (PFAS), a group of man-made chemicals, is an emerging threat to public health. They are toxic at extremely low levels (parts per trillion). They are persistent in the environment and remain in the human body for long periods of time. There is no way to remove them from the human body. There is widespread use of PFAS. It is used in water resistant clothing, stain resistant carpets, non-stick cookware and fire -fighting foam. The very qualities that make these chemicals resistant to stain and stick also make them a rapidly increasing environmental and public health hazard.

Major pathways of exposure include: 1) drinking contaminated water, 2) ingesting contaminated food (for ex., eating fish), 3) ingesting by hand to mouth activities (especially, children and babies) from surfaces treated with stain protectants, such as carpets and 4) working in a facility that manufactures or uses PFAS products.

The Agency for Toxic Substances and Disease Registry (part of CDC) lists the following impacts of PFAS on human health:

1. affect growth, learning, and behavior of infants and older children
2. lower a woman's chance of getting pregnant
3. interfere with the body's natural hormones
4. increase cholesterol levels
5. affect the immune system
6. increase the risk of cancer

Children, infants, and the developing fetus are at a far greater health risk because they drink more water in proportion to their weight, their brains and organs are developing rapidly, and they have a longer life to accumulate toxins. Infants fed formula mixed with water contaminated with PFAS are at the highest risk of exposure from water for two reasons. First, formula makes up 100% of their diet. Second, infants ingest a higher concentration of chemicals in relation to their small size.

In May 2016, US EPA issued a drinking water advisory for PFOA and PFOS at 70 parts per trillion(ppt), individually or combined. However, many authorities think that level is too high as appropriate public safety level. In Nov 2017, the Drinking Water Quality Institute (DWQI) recommended health based maximum contaminant level of 14 ppt. DWQI is an advisory to the New Jersey Dept of Environmental Protection. In turn, NJ has sets levels of 14 ppt for PFOA, 13 ppt for PFNA and 13 ppt for PFOS. Vermont has set 20 ppt for the entire class of PFAS. Massachusetts is in the process of setting a limit much lower than the current EPA level.

I recommend that Madison Water Utility Board take the necessary steps to ensure the public health and safety of the citizens of Madison, Wisconsin.

1) Consult with toxicologists in New Jersey, Vermont, and Massachusetts to better understand why they chose a lower level of acceptable limits for PFAS with the goal of setting a lower limit for Wisconsin (lower than 70 ppt)
2) Ensure that continued contamination is not occurring from fire training sites, airports and landfills.
3) Recommend use of filters to decrease the exposure of infants and children while we await setting limits.
4) Anticipate and plan for discharge of PFAS into storm sewers, waterways, and drinking water that can result from intense rain and flooding. Event similar to summer 2018 in Madison are expected to increase with climate change.

Elizabeth J. Neary, MD, MS
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Member, Council of Environmental Health, American Academy of Pediatrics
Wisconsin Champion, Pediatric Environmental Health Specialty Unit (PEHSU), Region 5, EPA
Executive Committee, Wisconsin Environmental Health Network

ASTDR statement from June 2018

https://www.atsdr.cdc.gov/toxprofiles/tp200.pdf

What our neighbors in Michigan are doing about PFAS:

https://www.michigan.gov/documents/pfasresponse/Science_Advisory_Board_Report_6412947.pdf

DATE: FEBRUARY 5, 2019

TO: MADISON WATER UTILITY BOARD

FROM: JIM POWELL

RE: PFAS IN DRINKING WATER SUPPLY

I am glad that the Water Utility is taking seriously potential PFAS contamination of its drinking water supply. To adequately investigate the problem and satisfactorily inform the public of the extent of the problem, possible responses and planned activities, I encourage the Board to direct the Utility general manager to take the following actions:

1. **Test all wells for at least 30 PFAS compounds using the lowest available detection limits**. PFAS is a widely used class of chemicals, so it would be very surprising that only the Air National Guard base at Truax is the only source of contamination in our community. Industries and landfills are the most common vectors, and offhand I can think of two wells—8 at Olbrich and 16 off Mineral Point Rd—that are adjacent to landfills. Test them. And PFAS travels far—up to several miles—so test all wells.

   This precautionary approach most likely will save ratepayers' money in the long run. As the Utility discovered with its Paterson Operating Center reconstruction, inadequate contaminant testing can lead to unplanned remediation costs that cannot be recouped through Wisconsin's regulatory process. Knowing the extent of contamination upfront will allow the Utility to better plan capital projects—such as filters and air strippers—and make PSC rate case proposals.

2. **Pursue all avenues—city, county, state and federal—to determine the extent of the PFAS contamination at Truax Field, the County airport, burn pits, old Town of Burke sewage treatment plant and Bridges Golf Course land--all currently owned by the County--as well as Starkweather Creek.** It is important to know how much PFAS is headed to well 15, as well as what's there right now. The Wisconsin Air National Guard, Dane County and City of Madison all have owned or leased various parcels in question, so liable parties must be identified and share remediation costs; all costs must not be passed onto water ratepayers. The polluters and property owners should be held responsible, not the people drinking water.

3. **Consider other states' efforts to test, understand, regulate and remediate PFAS contamination.** Since the state Departments of Natural Resources and Health Services are late to the game in creating PFAS contamination standards, Madison does not have a state regulatory process to guide them in protecting human health. Fortunately, other states, using the latest research, test results and regulatory tools, do have protective standards to guide utility actions to protect drinking water supplies, dramatically more protective than the EPA's for for just two of 3,000 PFAS compounds (which nearly everyone in the health and environmental fields believes is too lax). The general manager would do well not to tell ratepayers that he simply would wait for the DNR and DHS to act; the Water Utility itself must act now and take

steps that will assure the public that it is most concerned about drinking water quality, and not politics.

4. **Ask the Common Council to create a PFAS Task Force.** As the points above make clear, the complexity and extent of PFAS contamination may well affect drinking water across the city, but it's hardly just a utility issue. People, the environment and the economy may all be affected. Therefore, drawing on city resources through the creation of task force will best allow the water utility board to discharge its duties to protect and make safe the drinking water supply.