OBJECTIVE FOR

CANADIAN DRINKING WATER QUALITY

PER- AND POLYFLUOROALKYL SUBSTANCES

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Également disponible en français sous le titre :
Objectif pour la qualité de l'eau potable au Canada – Substances perfluoroalkylées et polyfluoroalkylées

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Publication date: August 2024

Cat.: H144-132/2024E-PDF
ISBN: 978-0-660-72710-3
Pub.: 240357

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INFORMATION ON OBJECTIVES FOR DRINKING WATER

This objective document was prepared in collaboration with the Federal-Provincial-Territorial Committee on Drinking Water.

The development of a guideline for the *Guidelines for Canadian Drinking Water Quality* follows a comprehensive scientific process and takes many years to finalize. In rare instances, new information on a contaminant raises concerns that need to be addressed more quickly than the guideline development process allows. In such cases, Health Canada, in collaboration with the provincial and territorial governments, may establish an objective to reduce exposure from drinking water while a guideline is revised or developed. This is the case with per- and polyfluoroalkyl substances (PFAS).

Drinking water objectives set out a goal for a maximum level of a contaminant in drinking water, taking into account available treatment technology and analytical methods. They are based on a review of scientific research and consider international regulatory information available at the time of their development. Drinking water objectives are externally peer reviewed to ensure scientific integrity.

The objective for PFAS in drinking water represents a precautionary group-based approach. The objective value of 30 ng/L (for the sum total of 25 specific PFAS) was established to reduce exposure to PFAS in drinking water. Where PFAS are detected in drinking water, the management of health risks and communication of the issue, as well as treatment options (if required) should be carefully examined with the stakeholders involved while considering the specifics of each situation.

OBJECTIVE VALUE

To reduce exposure from drinking water, an objective of 30 ng/L is established for the sum of the concentration of 25 per- and polyfluoroalkyl substances (PFAS) detected in drinking water. These 25 PFAS are (refer to Appendix A for full names):

PFBA
PFPeA
PFHxA
PFHpA
PFOA
PFNA
PFDA
PFUnA
PFDoA
PFBS
PFPeS
PFHxS
PFHpS
PFOS
4:2 FTS
6:2 FTS
8:2 FTS
HFPO-DA
ADONA
PFMPA
PFMBA
NFDHA
9Cl-PF3ONS
11Cl-PF3OUdS
PFEESA

When calculating the sum of PFAS for this objective, a result of “non-detect” is considered to have a value of zero. It is recommended that PFAS concentrations in drinking water be maintained as low as reasonably achievable (ALARA).

BACKGROUND

In 2018 and 2019, Health Canada established drinking water guidelines for perfluorooctane sulfonate (PFOS) and perfluorooctanoic acid (PFOA), and screening values for 9 other PFAS including perfluorobutanoate (PFBA), perfluoropentanoate (PFPeA), perfluorohexanoate (PFHxA), perfluoroheptanoate (PFHpA), perfluorononanoate (PFNA), perfluorobutane sulfonate (PFBS), perfluorohexanesulfonate (PFHxS), 6:2 fluorotelomer sulfonate (6:2 FTS) and 8:2 fluorotelomer sulfonate (8:2 FTS). These values applied to water intended for human consumption. Note that full names for any other PFAS mentioned in this document can be found in Appendix B.

On April 24, 2021, the Government of Canada published a Notice of Intent signalling its intent to move forward with activities that treat PFAS as a class. On May 20, 2023, the Government of Canada released the Draft State of Per- and Polyfluoroalkyl Substances (PFAS) Report which provides a qualitative assessment of the sources, fate, occurrence, and potential impacts of PFAS on the environment and human health to inform decision-making on PFAS as a class in Canada. In light of this assessment and considering new scientific evidence in the field of hazard and exposure assessment as well as in treatment and analytical technologies, a review of the current PFAS drinking water guidelines and screening values is under way.

This objective, based on the sum of specific PFAS detected, serves to reduce potential exposure to PFAS through drinking water while the reassessment of the guidelines and screening values is being completed. This objective replaces the two previous drinking water guidelines and nine screening values derived for individual PFAS. This technical document was prepared in collaboration with the Federal-Provincial-Territorial Committee on Drinking Water (CDW).

EXPOSURE CONSIDERATIONS

PFAS are a class of thousands of substances and, as defined by the Organisation for Economic Cooperation and Development (OECD) (2021), they include any chemical with at least a perfluorinated methyl group (–CF3) or a perfluorinated methylene group (–CF2–). The linked fluorine and carbon atoms in these substances produce a very stable molecule that is essentially unreactive and persists in the environment. Because of their properties, PFAS are used in a wide range of industrial processes and consumer products, such as surfactants and water and grease repellents. For example, PFAS are used in firefighting foams, textiles (including carpets and clothing), non-stick cookware, cosmetics and paper food packaging. In Canada, some of the legacy PFAS (that is, PFOA, PFOS and long-chain PFCAs, their salts and their precursors) have been prohibited from manufacture, use and import, with a limited number of exemptions. PFAS primarily enter Canada in products or as constituents of manufactured items. Some PFAS may be used in industrial processes, which may lead to releases from industrial facilities into the environment. PFAS releases from municipal solid waste landfills, municipal incineration, composting of PFAS-containing food packaging, wastewater treatment systems, and the application of biosolids to land are potential pathways of human and environmental exposure to PFAS (Guerra et al., 2014; Hamid et al., 2018; Lazcano et al., 2020; Liu et al., 2021a; Gewurtz et al., 2024). Many studies have demonstrated that PFAS can be transported long distances through the soil, water and air beyond the point at which they entered the environment. Due to their widespread use and persistence in the environment, PFAS can be found all over the world, in people, fish, wildlife and virtually every environmental compartment, including in remote areas.

The distribution of PFAS in the environment is dependent on the specific physical-chemical characteristics of the PFAS, including its chain-length, electrostatic charge, type of functional group and extent of fluorination (ITRC, 2023b). Ionic shorter chain perfluoroalkyl acids (PFAAs) tend to be more soluble in water and less absorbed to soil particles than the longer chain PFAAs. This allows them greater mobility in the aquatic environment. Three PFAS (PFHxA, PFBS, HFPO-DA) have been identified as being particularly mobile in water (ECHA, 2022b). Some PFAS, such as PFAAs, have a hydrophilic (water loving) head and a hydrophobic (water fearing) tail, which gives them surfactant-like properties and causes them to aggregate at interfaces (such as the water-air interface). It is due in part to these properties that PFAS have been used in aqueous film-forming foams (AFFFs) which form a thin film of water over a fuel source (ITRC, 2023b).

In Canada, PFAS contamination of the aquatic environment can result from both point and non-point sources. Non-point sources of PFAS may include surface runoff from urban areas and wet/dry atmospheric deposition (Lalonde and Garron, 2022). The most common point sources of PFAS contamination are associated with the use of firefighting foams containing PFAS. These foams, including AFFFs, are used during firefighting training activities and to extinguish fuel fires (for example, at airports and military bases) (D'Agostino and Mabury, 2017; Liu et al., 2021b). These AFFFs contain proprietary mixtures of PFAS and other chemicals (Liu et al., 2024). A number of PFAS have been detected in groundwater at airports and former fire-training areas where AFFFs have been used. PFAS have also been reported in ground and surface water at other types of sites (for example, emergency response locations, AFFF lagoons, hangarrelated storage tanks, firefighting equipment maintenance areas and pipelines or infrastructure impacted by AFFFs) (Awad et al., 2011; Anderson et al., 2016; Milley et al., 2018).

In addition to groundwater contamination, wells may be impacted by PFAS from various sources including domestic leach fields associated with septic systems (Schaider et al., 2016). Household wastewater moving through the leach fields may contain PFAS from food packaging, cookware, laundered clothing, and other household items (Müller et al., 2011).

People living in Canada can be exposed to PFAS through drinking water (see references below), food (Tittlemier et al., 2007; Aker et al., 2023), dust (Kubwabo et al., 2005; Shoeib et al., 2011; De Silva et al., 2012; Eriksson and Kärrman, 2015; Karaskova et al., 2016; Steeves et al., 2023) and indoor air (Shoeib et al., 2011; Beesoon et al., 2012). Drinking water can be a major source for exposure to PFAS in communities where the source water has been contaminated. However, for the general population (that is, those individuals not exposed occupationally or not living near point sources of contamination), the relative importance of each exposure source can vary substantially across different populations and by PFAS compound (Sunderland et al., 2019; De Silva et al., 2021). Furthermore, variable concentrations of PFAS in exposure media, differing assumptions about how long and how frequently people are exposed, and the consideration of PFAS precursors (which can degrade into more persistent PFAS) can all impact the calculation of the relative contribution of each exposure source (De Silva et al., 2021). Due to all these factors (as well as a general lack of data on PFAS in different exposure media) a general conclusion cannot be made on the relative importance of drinking water as a pathway for PFAS exposure for the general Canadian population.

Although currently there are limited data regarding PFAS in Canadian freshwater sources and drinking water, the body of evidence is growing. The number and suite of PFAS present in any given drinking water source will vary depending on the source of contamination, environmental conditions as well as new and historical patterns of use.

The Government of Canada carries out PFAS monitoring at freshwater sites across Canada. For example, from 2013 to 2020, 29 sites across Canada (located in every province except Alberta and Prince Edward Island) were sampled for 13 different PFAS to determine concentrations and trends. Sampling sites and frequencies varied during the study. PFAS were detected in the surface water of every province tested. Detection limits ranged from 0.4 to 1.6 ng/L. Of the 13 PFAS detected in 566 freshwater samples, PFBA, PFPeA, PFHxA, PFHpA, PFOA and PFOS had higher detection frequencies than the other PFAS. Within this group of PFAS, concentrations of PFBA and PFPeA increased significantly over the 2013 to 2020 period whereas concentrations of PFHpA, PFOA and PFOS decreased. The highest concentrations were noted to be 138 ng/L for PFBS (although this PFAS had fewer detections than the six PFAS identified above) and 137 ng/L for PFHxA. The authors note that this study found a higher frequency of detections of the replacement PFAS, such as PFBA, PFPeA, PFHxA, PFHpA and PFBS, than that seen in older Canadian studies (Lalonde and Garron, 2022).

PFAS have not been regularly monitored at drinking water treatment plants in Canada. Where monitoring data exists, it is often for a limited number of PFAS. Further, there is variability in the type of PFAS studied, the analytical methods used, the detection limits, the sampling frequency and the general study designs. Therefore, it is challenging to get an accurate picture of the concentrations of PFAS in drinking water across Canada. The information below includes a summary of data submitted by provinces as well as information from the scientific literature.

In Saskatchewan, the Water Security Agency collected drinking water samples (n = 7) from 7 water treatment plants in 2018–2019 to determine levels of PFOA and PFOS in treated drinking water. Neither PFOA nor PFOS were detected (method detection limit [MDL] of 2 ng/L) in the drinking water of 6 out of 7 communities. PFOA was detected in the single sample from one drinking water treatment plant at a concentration of 3 ng/L (Saskatchewan Water Security Agency, 2022). In 2023, the Water Security Agency once again collected and analyzed drinking water samples (n = 7) from 7 water treatment plants. A larger number of PFAS (up to 30 parameters) were included in the analysis, using methods with lower detection limits. At least one PFAS compound was detected in every sample. A maximum of 4 different PFAS compounds were detected in a few of the treated water samples. The highest concentration of summed PFAS compounds found in a sample (23 ng/L) was below the 30 ng/L objective.

Between 2012 and 2016, the Ontario Ministry of the Environment, Conservation and Parks measured the occurrence and concentrations of 14 PFAS (PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFUnA, PFDoA, PFBS, PFHxS, PFOS, PFDS and PFOSA) in 25 drinking water systems in Ontario (water intakes and treated drinking water). MDLs ranged from 0.5

to 1 ng/L, and results less than the MDL were substituted with a value of half the MDL (Kleywegt et al., 2020). PFUnA, PFDoA, PFDS and PFOSA were not detected in any drinking water samples. The most frequently detected compounds in Ontario drinking water were PFOA (73%; median 1.1 ng/L, maximum 6.6 ng/L), PFBA (67%; median 2.4 ng/L, maximum 10 ng/L), PFHxA (54%; median 1.3 ng/L, maximum 13 ng/L), PFPeA (51%; median 1.0 ng/L, maximum 15 ng/L) and PFOS (50%; median 0.63 ng/L, maximum 5.9 ng/L).

Similar median concentrations of PFBA, PFPeA, PFHxA, PFOA and PFOS were reported in samples of drinking water sourced from 19 sites around Lake Ontario and the St. Lawrence River (n = 8) and other lakes and small rivers in Canada (n = 11). Maximum concentrations of PFAS ranged from 0.1 ng/L (PFDA) to 4.1 ng/L (PFOS) in the Great Lakes–St. Lawrence samples, and 0.1 ng/L (PFUnA) to 4.9 ng/L (PFOA) for the rest of the Canadian tap water samples. PFHxA was detected in all Canadian tap water samples from this study. Other PFAS that were frequently detected included PFBA (95%), and PFHxS and PFOS (both 89%), while PFPeA, PFHpA, PFOA, PFNA, PFDA and PFBS were detected in at least 84% of the samples. Other PFAS detected in Canadian waters included PFOSA (53%), 6:2 FTSA (37%) and 5:3 FTCA (11%), as well as PFUnA, PFDoA and 7:3 FTCA, which were each detected in less than 10% of samples. A qualitative screening approach indicated that FBSA, FHxSA, PFECHS and PFPeS were occasionally present in tap water (semi-quantitative concentrations ranged from below the limit of detection to 1.2 ng/L), whereas PFEtS, PFPrS and PFPeS were below the limit of detection for all Canadian samples. The limits of detection for tap water ranged from 0.01 to 0.08 ng/L (Kaboré et al., 2018).

In a study that included 5 tap water samples from Niagara-on-the-Lake, Ontario, PFOA and PFOS were found at concentrations of 2.1 ng/L and 3.3 ng/L (arithmetic means). PFBA, PFPeA, PFHxA, PFHpA, PFNA, PFDA, PFUnA, PFHxS and PFEtS were also detected in the samples. The limits of quantification ranged from 0.004 to 1.6 ng/L (Mak et al., 2009).

At 7 sites in Quebec, source and treated water samples were collected monthly between April 2007 and March 2008. PFOA was detected in 75% of treated water samples (MDL of 0.3 to 0.6 ng/L), with a median value of 2.5 ng/L and a maximum value of 73.0 ng/L. PFOS was detected in 52% of treated samples (MDL of 0.3 to 0.6 ng/L), with a median value of 1.0 ng/L and a maximum value of 12.0 ng/L. PFNA and PFUnA were also detected in some samples (Berryman et al., 2012).

Between 2016 and 2021, Quebec's Ministère de l'Environnement, de la Lutte contre les changements climatiques, de la Faune et des Parcs (MELCCFP) sampled 41 drinking water treatment systems, testing for 18 PFAS (PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFUnA, PFBS, PFHxS, PFHpS, PFOS, PFDS, FHUEA, FOUEA, 4:2 FTS, 6:2 FTS, 8:2 FTS). Both surface and groundwater systems were included, with groundwater systems added in 2018

(MELCC, 2022). The sampling sites were selected based on previous PFAS detections and/or concerns in those locations, or due to their proximity to known potential point sources of PFAS. Detection limits ranged from 0.5 to 5 ng/L for raw water samples and from 0.3 to 5 ng/L for treated water samples. Among the 18 PFAS analyzed, 6 (PFPeA, PFHxA, PFHpA, PFOA, PFNA and PFOS) were detected in 10% or more of the samples taken. The 2016 data showed a reduction in the maximum concentrations of PFOA and PFOS (6 ng/L and 3 ng/L, respectively) when compared with the maximum surface water concentrations from the same sites sampled in 2007–2008 (66 ng/L for PFOA and 8.8 ng/L for PFOS). In the St. Lawrence River and other rivers, 5 substances (PFHxA, PFHpA, PFOA, PFNA and PFOS) were detected in at least 30% of the samples. PFOA and PFHxA were detected at the highest frequency (72% and 59%, respectively); both had a maximum concentration of 6 ng/L and a median concentration of 2 ng/L. In Lake Memphremagog, PFOA (median 1 ng/L, maximum 2 ng/L) and PFHxA (median 1.5 ng/L, maximum 3 ng/L) were detected in raw water; both were also found in treated drinking water at a maximum of 1 ng/L and median of 1 ng/L each. In groundwater sources, PFPeA (median 4 ng/L, maximum 48 ng/L) and PFHxA (median 3 ng/L, maximum 30 ng/L) were found in 14% and 17% of samples respectively, while PFOA (median 2 ng/L, maximum 4 ng/L) and PFOS (median 2 ng/L, maximum 3 ng/L) were found in 6% and 4% of samples (MELCC, 2022).

Munoz et al. (2023) conducted a study to validate a new analytical technique and to characterize PFAS in water samples across Quebec. Between 2018 and 2021, a total of 463 tap water samples were taken from within 376 municipalities in Quebec and examined for the presence of PFAS. Targeted analyses identified 31 PFAS in the water samples while an additional 23 PFAS were identified using nontarget screening. Individual detection limits ranged from 0.001 to 0.082 ng/L. 99.3% of the tap water samples contained PFAS and the concentrations of the total PFAS detected in each sample ranged from below the detection limit to 108 ng/L (median 2 ng/L, 95th percentile 13 ng/L). Only ten samples from five different localities had summed PFAS levels that were above 30 ng/L. The PFAS most frequently detected included PFOA (88%, median 0.27 ng/L, maximum 8.1 ng/L) and PFOS (80%, median 0.15 ng/L, maximum 13 ng/L). In addition, short-chain (C3-C6) perfluoroalkyl sulfonamides (FASA) were frequently detected (for example, FBSA detection = 50%) but at lower concentrations (< 1 ng/L). Of note, this study also reported the presence of emerging PFAS, such as 6:2 FTSAS-sulfone and 5:1:2 FtB, which were present at concentrations greater than 1 ng/L but were not widely detected. In addition, PFAS that have never been measured in drinking water before (for example, HO-X:2 FTS, FTSAS-related compounds, N-SPAmP-FASA-related compounds, TAmPr-FASA, X:3 FtB, and X:1:2 FtB) were detected in some of the samples. Overall, concentrations of PFAS were higher in tap water that had a surface water source as compared to tap water samples that had a groundwater source. However, of the top ten most contaminated locations, six had groundwater sources for the drinking water. The study by Munoz et al. (2023) was a comprehensive study that included many samples covering much of the province. It also measured a comparatively large number of PFAS often with a lower MDL than seen in other analytical methods.

In Nova Scotia, as of 2019, municipalities have been required to test the raw and treated drinking water for the presence of PFOA and PFOS. Neither PFOA nor PFOS have been detected in the 9 systems tested to date (NSECC, 2022). However, it is noted that the analytical method used in these analyses had a relatively high MDL of 20 ng/L.

In a 2023 pilot study involving 14 water treatment plants, the Government of Canada measured PFAS concentrations in paired source and treated water samples collected during the winter season. Samples were analyzed using a method developed to measure 38 PFAS compounds. The MDLs for this method ranged from 0.01 to 0.23 ng/L. The PFAS detected at the highest frequency in drinking water were PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFBS, PFPeS, PFHxS, PFHpS and PFOS. Median concentrations for individual PFAS, estimated from graphs, ranged from 0.05 to 0.80 ng/L for raw water and from 0.06 to 0.35 ng/L for treated water (Fan, 2023). Data from the study will be used to inform the development of drinking water guidelines.

In the United States (U.S.), nationally representative drinking water occurrence data were collected for 6 PFAS (PFHpA, PFOA, PFNA, PFBS, PFHxS, PFOS) under the Third Unregulated Contaminant Monitoring Rule (UCMR3). The minimum reporting levels (MRLs) established by the U.S. EPA for the monitored compounds ranged from 10 to 90 ng/L and were generally higher than the limit of quantitation of most published studies (Hu et al., 2016). The data collected between 2013 and 2015 revealed that 1.6% of the 36 977 samples and 4% of the 4 920 public water systems contained at least one detectable PFAS compound (Guelfo and Adamson, 2018). PFOA (MRL 20 ng/L) and PFOS (MRL 40 ng/L) were detected most frequently across all system sizes and source types at 1.03% and 0.79%, respectively. The highest maximum concentrations of PFOS (7 000 ng/L), PFHxS (1 600 ng/L), PFHpA (410 ng/L), PFOA (349 ng/L) and PFNA (56 ng/L) were detected in large systems with a groundwater source. PFBS (MRL 90 ng/L) was detected only in large systems, and the highest maximum concentration of 370 ng/L was observed in a large system supplied by a surface water source (Crone et al., 2019). An analysis of the UCMR3 data found that approximately 50% of samples with PFAS detections contained 2 or more PFAS and 72% of detections occurred in groundwater (Guelfo and Adamson, 2018). Certain activities, including the number of industrial sites that manufacture or use PFAS compounds, the number of military fire training areas and the number of wastewater treatment plants, were significant predictors of PFAS detection frequencies and concentrations in public water supplies (Hu et al., 2016).

Under the fifth cycle of the UCMR (UCMR5), 29 PFAS are scheduled to be sampled between 2023 and 2025. MRLs for individual substances were established by the U.S. EPA for this survey and range between 2 and 8 ng/L, except for NFDHA which has an MRL of 20 ng/L. Preliminary data, representing approximately 15% of the total results, shows that, of the 29 PFAS, 18 substances were measured at or above their MRL by at least one water system. In the approximately 3 000 water systems that were sampled, PFPeA (14%), PFBA (14%), PFHxA (13%), and PFBS (12%) were the most frequently detected substances. PFOA and PFOS were detected in 9.5% and 10.7% of systems, respectively. The lower MRLs in UCMR5 may account in part for the higher detection frequencies of these substances as compared to UCMR3.

HEALTH CONSIDERATIONS

Due to the widespread use and persistence of PFAS, and the fact that many PFAS are mobile and can accumulate, PFAS have become ubiquitous in the environment. Certain PFAS can be found in humans, wildlife and almost all environmental compartments worldwide (Wang et al., 2017).

In humans, some well-studied PFAS, such as the perfluorocarboxylic and sulfonic acids (PFCAs and PFSAs), have been shown to be readily absorbed in the body and bind to proteins in the blood which serve as the primary transport mechanism (Kudo, 2015; Forsthuber et al., 2020). Once distributed throughout the body, these substances accumulate in the blood and well-perfused tissues such as the liver and kidneys (Kudo, 2015). Some of these substances can cross the placental barrier, resulting in potential in utero exposure to the developing fetus (Mamsen et al., 2019; Wang et al., 2019; Li et al., 2020; McAdam et al., 2023). They can also be transferred to infants and children via human milk (VanNoy et al., 2018; Rawn et al., 2022; Zheng et al., 2022). Many PFAS, including PFCAs and PFSAs, are not metabolized in the body, likely because of their high stability and the low reactivity of the carbon-fluorine bonds (ATSDR, 2021). However, precursors such as fluorotelomer alcohols (FTOHs) and polyfluoroalkyl phosphate esters (PAPs) can be biotransformed to several metabolites including PFCAs and PFSAs, which do not degrade further (Butt et al., 2014). Available data show that certain PFAS are eliminated very slowly from the body, likely due to their strong protein binding affinity and internal reabsorption processes (Yang et al., 2010; EFSA, 2020). As such, some PFAS (for example, PFOA, PFNA, PFHxS and PFOS) can accumulate and persist in the body for years (ATSDR, 2021). Other shorter chain PFAS (such as PFBA and PFHxA) are eliminated more quickly with estimated half-lives ranging from several days to several weeks (Chang et al., 2008; Russell et al., 2015).

Toxicological (in vitro and in vivo) and epidemiological information is available for only a limited number of PFAS. A number of international agencies and scientific publications have reviewed the human health hazards associated with these PFAS (for example, EFSA, 2020; ATSDR, 2021; Fenton et al., 2021; ECHA, 2022a, 2022b, 2023; NASEM, 2022; Polcher et al., 2023). The most comprehensively studied PFAS are the PFCAs (including PFOA) and PFSAs (including PFOS). Limited information exists for several other types of PFAS (for example ether PFAS, FASA and fluorotelomer-based substances), while no data are available on the vast majority of PFAS (Pelch et al., 2021; Sanexen, 2022a).

Based on the available information, it is evident that exposure to certain PFAS has the potential to affect multiple systems and organs (ATSDR, 2021). Further, many PFAS have been shown to affect the same organs and systems. Effects commonly reported in animal studies include effects on the liver, immune system, kidney, reproduction, development, endocrine system (thyroid), the nervous system, and metabolism (lipids, glucose homeostasis, body weight). Outcomes of human epidemiological studies involve similar endpoints (Fenton et al., 2021; Sanexen, 2022a) and there appears to be a general consensus that currently the evidence is most consistent for effects on serum lipids, liver enzymes, vaccine response and fetal growth (ATSDR, 2021; ITRC, 2023c). In addition, the International Agency for Research on Cancer has classified PFOA as carcinogenic to humans (Group 1) and PFOS as a possible human carcinogen (Group 2B) (Zahm et al., 2023). The California EPA has also added PFOS to its list of chemicals known to cause cancer (OEHHA, 2021). Several of the effects noted above have been observed even at exposure levels found within the general population (that is, without exposure to a specific contamination source) (ITRC, 2023d).

New information on well-studied PFAS shows effects at lower levels than previous studies (for example, HFPO-DA and its ammonium salt [U.S. EPA, 2021a]) and effects on new endpoints of concern that were not previously considered (for example, impacts on the gut microbiome [Lamichhane et al., 2023]).

Currently, only a small number of PFAS are monitored in human biomonitoring surveys both in Canada and internationally. Canadian biomonitoring surveys show that some PFAS (for example, PFOA, PFOS, and PFHxS) are present in almost 100% of people living in Canada despite risk management measures being in place in Canada for several years. Other PFAS (for example, PFDA and PFUnA) are commonly detected in over 50% of the population (Health Canada, 2021a). When considering Indigenous populations, levels of PFOA, PFHxS and PFOS were lower in the First Nations on-reserve population (2011 data) as compared to the general Canadian population (2009–2011 data), while levels of PFNA and PFDA were similar between the two populations (Health Canada, 2023). However, certain populations including children/youth from specific Anishinabe communities were found to have PFNA levels that were substantially higher than comparable groups in the general Canadian population (Caron-Beaudoin et al., 2019; Lemire et al., 2019; Dubeau et al., 2022). In addition, youth, pregnant women and adults from communities in Nunavik had levels of PFNA, PFDA, PFUnA and PFOS that were substantially higher than comparable groups in the general Canadian population (Caron-Beaudoin et al., 2020; Aker et al., 2023). In areas where drinking water is an important exposure source for PFAS, studies indicate that the treatment of drinking water to reduce PFAS levels can also reduce serum PFAS levels and possibly adverse health outcomes (Herrick et al., 2017; Waterfield et al., 2020).

People living in Canada are exposed to multiple PFAS simultaneously as can be seen from biomonitoring data (Health Canada, 2021a). Given the combined exposure to multiple PFAS and the similarity of affected endpoints, there are concerns that exposure to PFAS could be associated with cumulative effects (ECHA, 2022a). However, the hazards of exposure to PFAS mixtures are largely unknown. A limited number of in vitro and in vivo mixture studies have found that PFAS may have antagonistic, synergistic or additive effects depending on the animal test species, dose level, dose ratio, and mixture components (Ojo et al., 2021; Addicks et al., 2023).

ANALYTICAL CONSIDERATIONS

Analytical methods

Two validated standardized methods are currently available for the quantitation of PFAS compounds in drinking water (U.S. EPA, 2020a, 2020b). It is important that the analytical methods selected provide quantitative results for the 25 PFAS specified in this objective. The selected method should conform to current best practices for accurate testing of PFAS in aqueous media (for example, isotope dilution analysis and weak anion exchange [WAX] cleanup). The methods should also be sensitive and practical for application in commercial laboratories and meet the MRLs specified by the responsible authorities. For example, under UCMR5, MRLs ranging from 2 to 20 ng/L have been established (U.S. EPA, 2021b) for the analysis of PFAS in drinking water using U.S. EPA-approved methods (see Appendix A). Many laboratories can accurately report at 2 ng/L for most PFAS and 5 ng/L for the rest (U.S. EPA, 2021b). Clients should confirm with laboratories that they can reliably measure PFAS in drinking water at or below the MRLs established by the responsible authority. Table 1 summarizes the standardized methods that have been established by the U.S. EPA for measuring PFAS in drinking water or in aqueous matrices (wastewater, surface water, and groundwater).

The U.S. EPA has approved Methods 533 and 537.1 for use in drinking water. Of these two methods, Heath Canada recommends the use of Method 533, as it follows current best practices (such as isotope dilution) while 537.1 does not. Although Method 1633 is not approved by the U.S. EPA for analyzing drinking water, it is included in Table 1 as it has undergone a multi-laboratory validation study for the determination of specified PFAS in aqueous samples such as surface water and groundwater. Investigations were conducted to assess the effect of free chlorine and three chlorine quenching or buffering agents (trizma, ammonium acetate and thiosulphate) on the method's performance. It was determined that the presence of chlorine affects some of the PFAS (sulfonamide precursors) targeted by Method 1633; and that among the quenching agents tested, thiosulphate performed best in ensuring that specifications were met for all PFAS measured by the method (SGS Canada, 2024). Method 1633 is "performance-based," which means that modifications may be made without additional U.S. EPA review to improve performance (for example, to overcome interferences, or improve the sensitivity, accuracy, or precision of the results) provided that all performance criteria in the method are met. The responsible jurisdiction could choose to approve the use of Method 1633 for drinking water or validate and approve an alternate

Table 1. Standardized U.S. EPA methods for the analysis of PFAS in water

Method (Reference)	Methodology	Comments
EPA Method 533 (U.S. EPA, 2020b)	Isotope-dilution/anion exchange SPE/LC-MS/MS method for the determination of select PFAS in drinking water.	Measures 25 specific PFAS (none with greater than 12 carbons), including perfluorinated acids, sulfonates, fluorotelomers and poly/perfluorinated ether carboxylic acids. Requires the use of LC-MS/MS in MRM mode.
EPA Method 537.1 (U.S. EPA, 2020a)	Hydrophobic SPE/LC-MS/MS method for the determination of select PFAS in drinking water.	Measures 18 specific PFAS, including perfluorocarboxylic acids containing up to 14 carbons. The concentration technique relies on hydrophobic interactions, and as such is not suitable for the more hydrophilic shorter carbon chain PFAS, such as PFBA and PFPeA. Does not use isotope dilution analysis.
EPA Method 1633 (U.S. EPA, 2024a)	Isotope-dilution/SPE / LC-MS/MS method for the determination of select PFAS in aqueous, solid, biosolid, and tissue samples.	Measures up to 40 specific PFAS in the aqueous matrices (wastewater, surface water, and groundwater). Requires the use of LC-MS/MS in MRM mode. Does not include a quenching step. Performance-based.

SPE — solid-phase extraction; LC-MS/MS — liquid chromatography/tandem mass spectrometry; MRM — multiple-reaction-monitoring mode

analytical method (including establishment of MRLs) that measures the 25 PFAS in drinking water noted in this objective. It is recommended that any approved methods meet, at a minimum, the MRLs established in the UCMR5 for each substance (see Appendix A). The 25 PFAS included in the objective sum are a good reflection of the PFAS observed in Canadian data as noted in the Exposure Considerations section. However, new analytical methods measuring a greater number of PFAS are currently under development in many jurisdictions. Thus, it is possible that the profile and number of PFAS detected in drinking water will change over time. Health Canada will continue to monitor new research and recommend any changes necessary.

Although some methods measure a greater number of PFAS, only the 25 PFAS listed in the objective statement should be summed for comparison with the objective. Any value above the established MRL should be included in the summation of PFAS, with a value of zero assigned for any value below the reporting limit or for "non-detect" results. If the laboratory can make quality measurements below the MRLs established by the responsible authority, these lower measurements should be summed for the comparison with the objective. Where possible, utilities should consider analyzing a greater number of

PFAS than just the 25 listed in the objective (for example, using U.S. EPA Method 1633) to gain a better understanding of the PFAS present in the drinking water. This may better inform the selection of treatment that will reduce exposure to the greatest extent possible. Site characterization to better understand the potential sources of PFAS to the source water is also useful; guidance on site characterization is available elsewhere (U.S. EPA, 2022a; ITRC, 2023a, 2023e).

Screening methods

PFAS precursors can degrade to PFAAs under the right environmental conditions. The Total Oxidizable Precursors (TOP) assay oxidizes PFAS precursors into their corresponding PFAAs, which can then be measured using the EPA methods or other methodologies. It is a useful screening tool that can help provide a better understanding of the amount of PFAS in a sample including unknown precursor species that might otherwise be missed (that is, total PFAS load). Because the TOP assay does not identify individual precursors, data typically are reported as the net change in PFAA concentrations before and after oxidation (Rodowa et al., 2020). The TOP assay may under-quantify the precursors of ultrashort-chain PFCAs (C2 – C3) that are not currently covered by available LC-MS/MS analytical methods (Ateia et al., 2023). Best practices and limitations of the TOP assay are further described in Ateia et al. (2023) and ITRC (2023f).

Another common surrogate analysis for PFAS is the Total Organic Fluorine (TOF) analysis, which can be used for drinking water. While the TOF analysis can be useful, it is indiscriminate and may capture fluorine from non-PFAS compounds. To date, there has been no demonstrated method that avoids having to employ sample preparation steps and losing a portion of the TOF. The U.S. EPA (2024b) has released an Adsorbable Organic Fluorine (AOF) method (EPA Method 1621) for aqueous matrices that uses carbon adsorption to prepare the sample for the fluoride wash and the final combustion ion chromatography process. However, the application to drinking water is limited because the minimum detection limit is well above concentrations typically seen in drinking water sources.

The TOP assay and TOF analysis can more comprehensively assess the concentration of PFAS beyond those listed in the above methods. However, they are qualitative techniques that are not yet been standardized or undergone multi-laboratory validation. Despite their limitations, these assays can provide a better understanding of which PFAS are or may be present in water and their impact on the treatment system's operations. During treatment, these additional PFAS may break through more rapidly, necessitating more frequent media change-out or regeneration. Data from these assays could be used to augment the data from quantitative methods. Further information on sampling and analysis, including limitations and best practices for sample collection, for qualitative and quantitative methods is available in ITRC (2023f, 2023g).

TREATMENT CONSIDERATIONS

The source-to-tap or water safety plan approach, which includes careful selection of the highest quality water source and source water protection, is an accepted approach to manage risks to drinking water safety (O'Connor, 2002; CCME, 2004; WHO, 2012). All water utilities should implement a comprehensive, up-to-date, risk management water safety plan. These approaches require a system assessment to characterize the source water; describe the treatment barriers that prevent or reduce contamination; identify the conditions that can result in contamination; and implement control measures. A vulnerability assessment should be undertaken to identify hazards, including potential sources of contamination and susceptibility of the source water to PFAS contamination (Health Canada, 2021b). Characterization of the source water is necessary to evaluate the presence, identity and concentration of PFAS; if treatment for PFAS is needed, this information is essential to establish design parameters as well as operational conditions for the selected treatment. Source water control options, such as a change in water source or blending of source waters, can be considered for reducing elevated PFAS concentrations in drinking water. Characterization of any alternative source water before blending is critical to ensure that overall treated water quality is maintained.

Municipal water treatment

Typically, treatment efficacy studies are carried out with a limited suite of PFAS at concentrations much higher than those observed in raw and treated drinking waters (Crone et al., 2019). Removal efficacies may differ greatly for PFAS with different physicochemical properties (for example, carbon chain length) when evaluated at concentrations relevant to drinking water. The key issues to consider when selecting treatment technologies for PFAS removal are the presence of competing anions and PFAS species, organic matter and the frequency of regeneration or replacement required for the sorptive media used (Appleman et al., 2013). The effectiveness of drinking water treatment for PFAS removal will depend on several factors, including source water characteristics, concentration and type of PFAS, treatment goals and proper operation of the system at all times. Disposal or manipulation of sorptive media, concentrates or residuals is an important consideration when selecting a treatment technology for PFAS removal. For example, spent filtration (such as powdered activated carbon [PAC] and granular activated carbon [GAC]) and ion-exchange media will require specialized disposal, such as high-temperature reactivation/destruction, to avoid release of PFAS back into the environment. Similarly, membrane technologies generate PFAS liquid residuals (for example, membrane reject water) that will require treatment and/or specialized disposal (U.S. EPA, 2022b). The destruction of PFAS (for example, in treatment residuals or exhausted media) is a complex and rapidly growing area of research. Most PFAS destructive technologies are in the developmental stage and thus it is challenging to fully evaluate their efficacy and cost for the drinking water context. However, the results are promising for achieving the destruction of PFAS compounds (Berg et al., 2022; Meegoda et al., 2022). Existing and emerging PFAS destruction technologies include electrochemical oxidation, thermal degradation, supercritical water oxidation, plasma-based treatment, and sonolysis (Meegoda et al., 2022). The availability of disposal options for treatment residuals (including media), as well as the disposal requirements of the relevant authorities, may limit which treatment technology can be selected.

Common drinking water treatment technologies (for example, coagulation, flocculation and oxidation) are not effective for PFAS removal. While there are treatment technologies that can effectively remove certain PFAS, no single treatment can remove a wide range of PFAS under all conditions. The most effective treatment technologies (> 90% removal efficiencies for certain PFAS) are GAC, membrane filtration (reverse osmosis [RO] and nanofiltration [NF]) and anion exchange (AIX) (Appleman et al., 2013, 2014; Dickenson and Higgins, 2016; Sanexen, 2022b). Each treatment technology has advantages and disadvantages. Achieving the objective may require a treatment train that includes more than one technology, or a technology used multiple times in series to treat the suite of PFAS present in the raw water. To ensure continued and effective removal, each facility should establish operational conditions and parameters based on the selected treatment technology(ies) and the characteristics of the raw water, including PFAS type, concentration and treatment goals.

GAC technology has the most field-relevant data at full- and pilot-scale (Sanexen, 2022b), and has proven to effectively remove PFAS from drinking water at relatively low concentrations (Appleman et al., 2014). Additionally, GAC can maintain its performance across a broad range of water chemistries. However, GAC has demonstrated greater affinity for PFAS with chain lengths greater than 6 carbons compared with shorter chain PFAS (Gagliano et al., 2020). In addition, perfluorinated sulfonates are adsorbed more easily by GAC than their carboxylic acid counterparts due to their higher hydrophobicity (Du et al., 2014). As a result, increased frequency of GAC regeneration or replacement will be required when treating certain PFAS (Rodowa et al., 2020). Operational parameters such as GAC type (for example, bituminous coal), bed size and hydraulic loading rate also influence filter run time (Belkouteb et al., 2020). Determining breakthrough time is critical for ensuring PFAS removal continues to meet the treatment objective.

AIX resin properties, such as porosity, functional group and polymer matrix, influence PFAS treatment efficacy (Gagliano et al., 2020). Given many PFAS exist as anions at drinking water pH, strong base AIX resins are capable of removing these PFAS species (Crone et al., 2019). The AIX process also preferentially removes longer chain PFAS and perfluorosulfonates (Appleman et al., 2014). However, adjustments to AIX resin characteristics (for example, hydrophobicity of functional group) can increase the sorption capacity for less hydrophobic PFAS (Chularueangaksorn et al., 2014; Zaggia et al., 2016). Although AIX resins have the advantage of greater adsorption capacity than GAC, they are typically limited to a single use for drinking water applications (Ross et al., 2018; Crone et al., 2019). AIX resin regeneration has been achieved in some studies (Crone et al., 2019), albeit utilizing complex or unconventional procedures.

Membrane technologies such as RO and NF are both highly effective for removal of many PFAS. RO effectively removes PFAS of all chain lengths as a function of size exclusion and charge rejection; NF relies principally on electrostatic repulsion and hydrophobicity, particularly for removal of shorter chain PFAS (Dickenson and Higgins, 2016; Zeng et al., 2017). The degree of RO and NF rejection may vary among PFAS and may be substantially lower for chargeneutral PFAS such as PFOSA (Steinle-Darling and Reinhard, 2008; Steinle-Darling et al., 2010; Sanexen, 2022b). Fouling and scaling problems limit the wide-scale application of membrane technologies such as RO and NF.

A limited number of bench-scale studies have evaluated the removal of perfluoroalkyl acids (7 PFCAs and 3 PFSAs) by PAC. Based on those study results, median removal efficiency for individual PFAS by PAC was 64.5% and 79.3% for the sum of PFAS (Sanexen, 2022b). However, the removal efficiency is dependent on PFAS type and may be under- or overrepresented with a different list of PFAS. Due to inefficiencies, PAC needs to be combined with other treatment technologies (for example, PAC and coagulation processes) to achieve a removal rate of 90% or more (Bao et al., 2014; Pramanik et al., 2015). Pramanik et al. (2015) demonstrated removal efficiencies of 95% and 91% for PFOA and PFOS, respectively, for combined treatment with PAC before polyaluminium chloride coagulation. Such combined treatment processes can reduce PFAS exposure without major investments for utilities that have existing installations for coagulants and PAC addition. Additionally, this treatment approach may be a useful interim PFAS reduction solution before the installation of more advanced treatment technologies. The increase in PAC dosage and contact time can also increase PAC removal efficiencies of PFAS (Qu et al., 2009; Bao et al., 2014; Pramanik et al., 2015). Although encouraging, these bench scale results are available only for limited number of PFAS species. Increased dissolved organic carbon (DOC) concentrations in raw water may have a negative impact on PAC removal efficacy for PFAS. Pramanik et al. (2015) found that higher DOC concentrations in the feedwater had a greater impact on PAC than on GAC performance when removal of PFOA and PFOS was evaluated. The safe disposal of settled sludge containing PFAS-laden PAC is an important consideration.

Treatment achievability

Studies assessing pilot- and full-scale PFAS treatment achievability have demonstrated that GAC, AIX and RO can each effectively reduce concentrations of PFCAs and PFSAs in raw water (Sanexen, 2022b). In general, these studies were conducted with PFAS concentrations higher than those observed in Canadian source water, therefore some uncertainties exist with respect to treatment efficacy of each technology at lower PFAS concentrations. Additionally, site-specific conditions can differ significantly, affecting the treatment efficacy of the technologies and necessitating system-specific treatment strategies. For example, the PFAS treatment efficacy of GAC and AIX can be greatly affected by competitive ions such as natural organic matter, sulfate, chloride and bicarbonate in source water. Therefore, source water characterization should be part of routine system assessments. Understanding water quality parameters based on seasonal, temporal and climate change is important in managing treatment operations. To achieve the objective, treatment systems must be configured and operated properly which may lead to challenging operating conditions (such as very long empty bed contact times or frequent media regeneration or replacement) and may not be practically or economically feasible for some water treatment facilities.

Residential-scale water treatment technologies

In cases where an individual household obtains their drinking water from a private well that is impacted by PFAS, a drinking water treatment device may reduce the concentration of a limited number of PFAS in drinking water. Systems classified as residential scale may have a rated capacity to treat volumes greater than that needed for a single residence, and thus may also be used in small systems. Before a treatment unit is installed, the water should be tested to determine the general water chemistry and PFAS concentration and speciation in the source water. Ideally, periodic testing by an accredited laboratory should be conducted on both the water entering the treatment unit and the treated water to verify that the treatment unit is effective. Units can lose removal capacity through use and time and need to be maintained and/or replaced. It is important to follow the manufacturer's recommendations for operation and maintenance of a treatment device (for example, replacement of filter media).

Treatment devices can be certified to NSF Standard 53 (GAC) and NSF Standard 58 (RO) (NSF International, 2022a, 2022b) for the reduction of 7 PFAS from drinking water to less than or equal to a total concentration of 20 ng/L. The PFAS criteria is applicable for the total of the following 7 PFAS: PFHpA, PFOA, PFNA, PFDA, PFBS, PFHxS and PFOS. Certification for the removal of the following PFAS is also possible under these two standards: PFHpA, PFOA, PFNA, PFHxS and PFOS. The use of treatment devices certified to these criteria, even if certification only includes a limited number of PFAS, will help homeowners further reduce their exposure to PFAS in drinking water. Work is ongoing under these standards to integrate the maximum contaminant levels (MCLs) established under the U.S. EPA's final rule for six PFAS in drinking water: PFOA, PFOS, PFBS, PFHxS, PFNA, and HFPO-DA (U.S. EPA, 2024c). The MCLs are further described in the International Considerations section.

Ion exchange is not included in the NSF standards and therefore no systems are currently certified specifically for PFAS reduction. However, ion exchange systems with PFAS selective resins are expected to be effective for PFAS removal at the residential scale. Traditionally, ion exchange treatment systems have been installed at the point of entry (POE) of a building or home and are typically designed and constructed for use at the small system or residential-scale by drinking water system providers or dealers. If the selected drinking water treatment system is not certified for PFAS removal, it is strongly recommended to ensure that the system's materials and components be certified to NSF/ANSI/CAN Standard 61 (for leaching) (NSF International, 2021a). Any chemicals used in the treatment system should be certified to NSF/ANSI/CAN Standard 60 (additives) (NSF International, 2021b). These standards ensure that these systems are safe for use in drinking water applications.

Herkert et al. (2020) evaluated the effectiveness of point-of-use (POU) and POE residential drinking water filters in removing a suite of 16 PFAS. Various POU systems (for example, pitcher, refrigerator and faucet-mounted carbon-based filters; carbon block and RO under-sink systems) and some POE systems were included in the study. POE systems were either GAC-based or had both GAC and ion exchange media incorporated. All under-sink dual-stage carbon filters and RO systems showed a ≥ 90% removal efficiency for all PFAS evaluated with the exception of GenX (> 74% removal efficiency). All other POU filters containing activated carbon exhibited variable PFAS removal with long-chain PFAS (~60–70% removal) being more efficiently removed than short-chain PFAS (~40% removal). For the POE systems examined, PFAS removal was largely inconsistent and unpredictable. Increased PFAS levels were observed in the filtered water of some POE systems suggesting saturation of the media and PFAS desorption into filtered water.

When choosing a drinking water treatment device, special considerations should be given to the PFAS waste generated by the treatment. For example, RO residential treatment devices generate liquid waste that is discharged to a sewer or septic system, while spent, contaminated GAC filters may be sent to the landfill. As a result, PFAS can be reintroduced back into the environment. Homeowners should consult with local authorities to determine if treatment or an alternate source of drinking water is recommended, as well as for the available options for the disposal of treatment devices, media and/or residuals that may contain elevated PFAS concentrations.

APPLICATION OF THE OBJECTIVE

Note: Specific guidance related to the implementation of drinking water objectives should be obtained from the appropriate drinking water authority in the affected jurisdiction.

The objective of 30 ng/L applies to the sum of 25 specified PFAS that are detected in drinking water. The sum is calculated by adding the concentrations of each of the PFAS detected at or above the MRL established by the jurisdiction or the laboratory (whichever is lower). While a jurisdiction may choose to measure more than 25 PFAS, the objective of 30 ng/L applies only to the sum of the 25 PFAS specified in this objective. Information on any additional PFAS (for example, types detected, concentration, detection frequencies) is valuable and can be used to get a better sense of the PFAS profile of the water sample.

Testing for PFAS is highly specialized and should be conducted by a laboratory that is accredited for this analysis.

Since the objective is based on analytical and treatment achievability, if someone is exposed to an amount of PFAS above the objective, it does not necessarily mean that health problems will occur. Whether or not health problems develop depends on how much, how often, and for how long an individual is exposed, as well as to which PFAS an individual is exposed. A person's specific attributes including their age, habits, and overall health can impact how their body will react to chemical exposures.

In case of an exceedance of the objective of 30 ng/L, the management of health risks and treatment options (if required) should be carefully investigated by the responsible jurisdiction in collaboration with partners and stakeholders while considering the specifics of the situation, and, if appropriate, followed by corrective actions. These actions could include, but are not limited to, resampling, assessing the profile of the PFAS detected, protecting or changing source waters and evaluating water treatment strategies such as selecting or combining different treatment technologies based on water quality and resampling results. The objective represents a benchmark for all jurisdictions to strive towards. However, achieving it may take time given the technical complexity and cost of measuring and managing PFAS.

Given the potential for exposure to multiple PFAS at the same time, the potential for negative health impacts, and the limited data on many PFAS, if treatment is required, it is recommended that PFAS levels be maintained ALARA.

If measurements of PFAS in drinking water are approaching or exceed the 30 ng/L objective, it may be useful to examine the types of PFAS that are present in the greatest concentrations. For example, PFOA, PFOS, PFHxS and PFNA are currently considered to be among the most toxicologically potent PFAS (Polcher et al., 2023). If the PFAS profile shows that these substances are present in the highest concentrations, significant mitigative actions (such as treatment) should be considered as a precautionary measure. Of note, the Public Health Expertise and Reference Centre in Quebec have developed flowcharts to assist in decision making when PFAS are detected in drinking water (INSPQ, 2023; Ponce et al., 2024).

Utilities should characterize their source water to assess PFAS concentrations, particularly if source waters are impacted by firefighting training areas, military bases, airports, manufacturing sites and/or waste disposal sites. If PFAS is detected in treated drinking water, the summed value of the 25 PFAS should not exceed 30 ng/L. If treatment is required, the source should be sampled in conjunction with compliance monitoring. Samples should be collected after treatment, but prior to distribution, typically at the entry point to the distribution system. Paired samples of source and treated water should be taken to confirm the efficacy of the treatment.

Monitoring for PFAS should be conducted annually, at minimum. Utilities should consult with the responsible authority to determine the appropriate frequency of monitoring for their particular situation. Utilities that have baseline data showing the absence of PFAS may conduct less frequent monitoring in consultation with the responsible authority.

In cases where an individual household obtains their drinking water from a private well that is impacted by PFAS, a drinking water treatment system may reduce the concentration of a limited number of PFAS in drinking water. Treatment devices can be certified to NSF Standard 53 (GAC) and NSF Standard 58 (RO) (NSF International, 2022a, 2022b) for the reduction of seven PFAS from drinking water to less than or equal to a total concentration of 20 ng/L (see list of PFAS in Residential-scale water treatment technologies section). The use of treatment devices certified to these criteria will help homeowners further reduce their exposure to PFAS in drinking water.

INTERNATIONAL CONSIDERATIONS

Other national and international organizations have guideline values for PFAS in drinking water. Jurisdictions such as the European Commission, Sweden, Denmark and some U.S. states, have established a single guideline value for a combination of PFAS in drinking water. For example, the council of the European Union has adopted a directive that includes limits of 100 ng/L for the sum of 20 PFAS and 500 ng/L for the sum of all PFAS in drinking water. Member states have until January 2026 to comply with the limits (EU, 2020). The state of Maine has set a limit of 20 ng/L for 6 PFAS either individually or combined (State of Maine, 2021). Denmark has a drinking water quality value of 2 ng/L for the sum of 4 PFAS (Danish Ministry of the Environment, 2023).

The U.S. EPA has established a National Primary Drinking Water Regulation with legally enforceable levels (MCLs) for six PFAS in drinking water: PFOA, PFOS, PFBS, PFHxS, PFNA, and HFPO-DA. The MCLs of 4 ng/L for each PFOA and PFOS are based on the practical quantitation level for these substances (that is, the lowest level that can be reliably measured within specified limits of precision and accuracy during routine laboratory operating conditions). PFNA, PFHxS, and HFPO-DA each have an MCL of 10 ng/L. In addition, the U.S. EPA has set a hazard index of less than one for mixtures containing two or more of PFNA, PFBS, PFHxS, and HFPO-DA. The hazard index is the sum of the hazard quotient (that is, the ratio between the concentration of PFAS in the water and the health-based value) for each PFAS. The U.S. EPA has also established health-based, maximum contaminant level goals (non-enforceable) of zero for both PFOA and PFOS and 10 ng/L for each of PFNA, PFHxS, and HFPO-DA (U.S. EPA, 2024b).

Although the World Health Organization had developed an initial background document for PFOA and PFOS in drinking water in 2022, the organization is expanding the review to include the occurrence and health effects of substances beyond PFOS and PFOA. The document will also be expanded to consider sources of exposure beyond drinking water (WHO, 2023).

RATIONALE

Given the potential for exposure to multiple PFAS at the same time, the potential for negative health impacts, the uncertainty and the limited data on many PFAS, a precautionary group-based approach to PFAS is warranted. The lower the levels of PFAS, the lower the risk to public health. As such, the objective for PFAS in drinking water is based on a sum of PFAS detected in drinking water. This sum should be calculated using the 25 PFAS listed in the objective, such that the sum of their concentrations does not exceed 30 ng/L. For the purposes of this objective, a result of non-detect is considered to have a value of zero. For the reasons noted, it is also recommended that treatment plants strive to maintain PFAS concentrations in drinking water ALARA.

The objective value of 30 ng/L was established to reduce exposure to PFAS in drinking water. The value was determined to be reasonably achievable by treatment, considering:

- published treatment data for pilot- and full-scale treatment plants with a focus on the removal efficacy reported for a limited suite of PFAS, at environmentally relevant concentrations, in a variety of water qualities (Sanexen, 2022b);
- the concentration of PFAS achieved at pilot- and full-scale for each of GAC, AIX and RO treatment technologies with raw water concentrations that are comparable to those generally observed in Canadian source waters and contaminated sites;
- reporting levels for PFAS for which a validated and recognized analytical method is available (U.S. EPA, 2020a, 2020b); and
- Canadian monitoring data (see Exposure considerations section).

The objective offers the Canadian drinking water sector an efficient approach for risk management by providing only one target value for this group of chemicals, thereby reducing exposure to PFAS and potential health risk.

A traditional health-based approach was not adopted to derive the objective in part due to the rapidly evolving science. Further, the science is complex, there is currently no consensus regarding the most sensitive health effects, and approaches to hazard and risk assessment are varied. Consequently, health-based values derived by various jurisdictions differ from each other. As more toxicity data are published, an increasing number of health effects are being associated with exposure to PFAS and toxicological reference values (TRVs) for these substances are being set at lower levels. Furthermore, people living in Canada are exposed to multiple PFAS simultaneously and the potential hazard associated with exposure to these mixtures is unknown. Consequently, a substance-by-substance assessment of the TRVs available for each PFAS is not a sustainable approach for managing PFAS in drinking water.

Many of the PFAS addressed under the objective include the most studied PFAS (that is, PFCAs and PFSAs) and PFAS that are currently considered to be among the most toxicologically potent (for example, PFOA, PFNA, PFHxS and PFOS) (Sanexen, 2022a; Polcher et al., 2023). This group also includes the most commonly detected PFAS in drinking water (for example, PFBA, PFPeA, PFHxA, PFOA and PFOS). While there are limited monitoring data for PFAS in Canadian drinking water, data from other jurisdictions show that many of these substances are found in water and other environmental media and indicate the potential for transfer or leaching into drinking water (Reade and Pelch, 2020).

The objective includes 25 PFAS which can be measured by the available validated standardized analytical methods, or a method validated by a jurisdiction. The intent is to ensure a reduction of exposure to PFAS that can be quantified, while still allowing some flexibility in which method to use. It can be difficult to predict which PFAS will be present in a drinking water sample. For this reason, where possible, utilities should strive to analyze a greater number of PFAS than those in the objective using validated methods to better understand the PFAS in the drinking water. This will better inform treatment selection and ultimately help reduce PFAS exposure to the greatest extent possible.

In adopting a group-based objective, several principal considerations were taken into account. The PFAS studied to date have been shown to be extremely persistent, mobile and difficult, if not impossible, to remove from the environment once released. These properties make exposure to PFAS inevitable and potentially continuous. The adverse effects associated with every individual PFAS are currently unknown. However, for well-studied PFAS (such as PFOA and PFOS), more adverse effects are coming to light identified at ever lower levels. For less well-studied PFAS, as more research is conducted, adverse effects are identified. Therefore, based on what is known about well-studied PFAS and the potential for other PFAS to behave similarly, there are potential human health concerns for the group of PFAS. Consequently, a precautionary group-based approach is warranted.

Acknowledging all the above considerations, this objective is deemed the most appropriate approach to reduce potential exposure to multiple PFAS through drinking water while the formal guidelines are being revised. The objective will remain in place up until new, updated Guidelines for Canadian Drinking Water Quality are published. The full revision of the guidelines for PFAS in drinking water will include a comprehensive review of new toxicological evidence (for example, TRVs) and risk assessments. The revision will also include a comprehensive review of new information on analytical and treatment methods for PFAS as well as new findings for the management of treatment residuals. Health Canada will continue to monitor new research and recommend any changes necessary.

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APPENDIX A

List of 25 PFAS and minimum reporting levels for analysis by U.S. EPA Methods 533, 537.1 and 1633[a]

Abbreviation and CASRN	PFAS (total number of carbons)	In U.S. EPA Method 533 (Y/N)	In U.S. EPA Method 537.1 (Y/N)	In U.S. EPA Method 1633 (Y/N)	UCMR5 MRL[b] ng/L (see note)
11Cl PF3OUdS 763051-92-9	11-Chloroeicosafluoro-3-oxaundecane-1-sulfonic acid (C10)	Y	Y	Y	5
9Cl-PF3ONS 756426-58-1	9-Chlorohexadecafluoro-3-oxanonane-1-sulfonic acid (C8)	Y	Y	Y	2
ADONA 919005-14-4	4,8-Dioxa-3H-perfluorononanoic acid (C7)	Y	Y	Y	3
HFPO-DA[c] 13252-13-6	Hexafluoropropylene oxide dimer acid (C6)	Y	Y	Y	5
PFBS 375-73-5	Perfluorobutanesulfonic acid (C4)	Y	Y	Y	3
PFDA 335-76-2	Perfluorodecanoic acid (C10)	Y	Y	Y	3
PFDoA 307-55-1	Perfluorododecanoic acid (C12)	Y	Y	Y	3
PFHpA 375-85-9	Perfluoroheptanoic acid (C7)	Y	Y	Y	3
PFHxS 355-46-4	Perfluorohexane sulfonic acid (C6)	Y	Y	Y	3
PFHxA 307-24-4	Perfluorohexanoic acid (C6)	Y	Y	Y	3
PFNA 375-95-1	Perfluorononanoic acid (C9)	Y	Y	Y	4
PFOS 1763-23-1	Perfluorooctane sulfonic acid (C8)	Y	Y	Y	4
PFOA 335-67-1	Perfluorooctanoic acid (C8)	Y	Y	Y	4

Abbreviation and CASRN	PFAS (total number of carbons)	In U.S. EPA Method 533 (Y/N)	In U.S. EPA Method 537.1 (Y/N)	In U.S. EPA Method 1633 (Y/N)	UCMR5 MRL[b] ng/L (see note)
PFUnA 2058-94-8	Perfluoroundecanoic acid (C11)	Y	Y	Y	2
NFDHA 151772-58-6	Nonafluoro-3,6-dioxaheptanoic acid (C5)	Y	N	Y	20
PFBA 375-22-4	Perfluorobutanoic acid (C4)	Y	N	Y	5
8:2 FTS 39108-34-4	1H,1H, 2H, 2H-Perfluorodecane sulfonic acid (C10)	Y	N	Y	5
PFEESA 113507-82-7	Perfluoro(2-ethoxyethane) sulfonic acid (C4)	Y	N	Y	3
PFHpS 375-92-8	Perfluoroheptanesulfonic acid (C7)	Y	N	Y	3
4:2 FTS 757124-72-4	1H,1H, 2H, 2H-Perfluorohexane sulfonic acid (C4)	Y	N	Y	3
PFMPA 377-73-1	Perfluoro-3-methoxypropanoic acid (C4)	Y	N	Y	4
PFMBA 863090-89-5	Perfluoro-4-methoxybutanoic acid (C5)	Y	N	Y	3
6:2 FTS 27619-97-2	1H,1H, 2H, 2H-Perfluorooctane sulfonic acid (C8)	Y	N	Y	5
PFPeA 2706-90-3	Perfluoropentanoic acid (C5)	Y	N	Y	3
PFPeS 2706-91-4	Perfluoropentanesulfonic acid (C5)	Y	N	Y	4

[a] For EPA Method 1633, only the 25 PFAS for the objective are listed for comparison to the total.

[b] MRL = Minimum reporting level established by the U.S. EPA for analyses under UCMR5 (U.S. EPA, 2021b)

[c] HFPO-DA and the ammonium salt of HFPO-DA are components of the GenX processing aid technology, and both are measured as the anion of HFPO-DA by this method.

Note: The UCMR MRL is defined as the lowest measurable concentration that, with 95% confidence, is achievable by at least 75% of laboratories participating in EPA's nationwide UCMR MRL-setting studies on PFAS, using a specified analytical method. Alternate MRLs may be established by a laboratory for their specific purpose or may be set by a jurisdiction.

Adapted from U.S. EPA (2021b)

APPENDIX B
LIST OF ABBREVIATIONS

(Note that this list includes only PFAS that are not listed and spelled out in Appendix A)

5:1:2 FtB	5:1:2 fluorotelomer betaine
5:3 FTCA	5:3 FTCA fluorotelomer carboxylic acid
6:2 FTSA	6:2 fluorotelomer sulfonic acid
6:2 FTSAS-sulfone	sulfonyl analogue of 6:2 fluorotelomer thioether amido sulfonate
7:3 FTCA	7:3 FTCA fluorotelomer carboxylic acid
AOF	adsorbable organic fluorine
AFFF	aqueous film-forming foams
AIX	anion exchange
ALARA	as low as reasonably achievable
DOC	dissolved organic carbon
FASA	perfluoroalkyl sulfonamide
FBSA	perfluorobutane sulfonamide
FHUEA	2H-perfluoro-2-octenoic acid
FHxSA	perfluorohexane sulfonamide
FOUEA	2H-perfluoro-2-decenoic acid
FTCA	fluorotelomer carboxylic acid
FTOHs	fluorotelomer alcohols
FTSA	fluorotelomer sulfonic acid
FTSAS	fluorotelomer thioether amido sulfonates
GAC	granular activated carbon
HO-X:2 FTS	hydroxy-X:2 fluorotelomer sulfonates
LC-MS/MS	liquid chromatography/tandem mass spectrometry
MCL	maximum contaminant level

MDL	method detection limit
MRL	minimum reporting level
MRM	multiple-reaction-monitoring
NF	nanofiltration
N-SPAmP-FASA	N-sulfopropyl dimethylammoniopropyl perfluoroalkyl sulfonamide
PAC	powdered activated carbon
PAPs	polyfluoroalkyl phosphate esters
PFAAs	perfluoroalkyl acids
PFAS	per- and polyfluoroalkyl substances
PFCAs	perfluorocarboxylic acids
PFSAs	perfluorosulfonic acids
PFDS	perfluorodecane sulfonic acid
PFECHS	perfluoroethylcyclohexane sulfonic acid
PFEtS	perfluoroethane sulfonic acid
PFOSA	perfluorooctane sulfonamide
PFPrS	perfluoropropane sulfonic acid
POE	point of entry
POU	point of use
RO	reverse osmosis
SPE	solid-phase extraction
TAmPr-FASA	trimethylammoniopropyl perfluoroalkyl sulfonamide
TOF	total organic fluorine
TOP	total oxidizable precursors
TRV	toxicological reference value
UCMR	Unregulated Contaminants Monitoring Rule
U.S. EPA	United States Environmental Protection Agency
WAX	weak anion exchange
X:1:2 FtB	X:1:2 fluorotelomer betaine
X:3 FtB	X:3 fluorotelomer betaine