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March 23, 2020

Leah J. Smith
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Florida Department of Environmental Protection
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Re: Update on PFAS criteria development in other states

Dear Ms. Smith:

At your request, we have reviewed the development of perfluoroalkyl (PFAS) criteria for drinking water and groundwater by the federal government and states. This document represents an attempt to summarize the current PFAS drinking water and groundwater criteria in the United States and the methods used to calculate them. The summary includes both promulgated values and values that are in various stages of an approval process, and is intended to facilitate comparison of approaches for deriving PFAS drinking water and groundwater criteria by various environmental agencies. We found that transparency in methods for deriving these criteria varied substantially. For some, there was thorough documentation from the source agency and clear explanations for choices made in approach and assumptions. For others, details regarding the basis for the values were not found or were obtained indirectly from secondary sources (e.g., documents from other agencies).

In 2016, the United States Environmental Protection Agency (USEPA) developed health advisory levels (HALs) for perfluorooctanoic acid (PFOA) and perfluorooctane sulfonic acid (PFOS) in drinking water. A number of states have adopted formally or informally these values (70 ng/L for PFOA, 70 ng/L for PFOS, and 70 ng/L for the sum of PFOA and PFOS) for assessment of PFAS contamination of drinking water and/or groundwater (Table 1). As noted in Table 1, Connecticut has applied the 70 ng/L limit to the sum of a number of specific PFAS beyond PFOA and PFOS, in effect assuming that the toxicity of these additional PFAS is similar to PFOA and PFOS and that their effects are additive.

In 2018, the Agency for Toxic Substances and Disease Registry (ATSDR) derived Minimal Risk Levels (MRLs) for PFOA and PFOS, and from these Environmental Media Evaluation Guidelines (EMEGs) for drinking water were developed, including values for children. The EMEGs for children were substantially lower than the USEPA HALs — 21 µg/L for PFOA and 14 µg/L for PFOS (Table 2). Several states have also derived their own criteria for PFOA

and PFOS that are different from USEPA HALs, as well as drinking water and/or groundwater criteria for a number of other PFAS. These criteria are summarized in Table 2. Like the ATSDR EMEGs, all of the PFOA and PFOS drinking water criteria developed by states except Nevada are lower than the USEPA HALs. Explanations for the differences in criteria from the USEPA HALs, and from each other, for most states can be found in Tables 3-5 for PFOA and 6-8 for PFOS. The proposed PFOA and PFOS criteria for Illinois are identical to the ATSDR child EMEGs, and it can be speculated that the intent is for Illinois to adopt the ATSDR values. However, we were unable to confirm this at the time of this report, so the basis for these values is not included in Tables 3-8. Also, the proposed New York criteria of 10 ng/L for PFOA and PFOS were not based directly on a specific approach or set of assumptions, but rather reflect a management approach given the range of options available. This approach is explained further later in this report, but for purposes of comparisons in Tables 3-8, New York is also not included.

All of the criteria in Table 2 are based upon non-cancer effects of the various PFAS. In addition to non-cancer effects, California also developed drinking water limits based upon carcinogenicity for PFOA and PFOS, which are lower (see footnote in Table 2). To facilitate comparison with other states, the criteria shown in Table 2 for California are their non-cancer numbers. Criteria based upon carcinogenicity are discussed later in this report.

PFOA

The critical effects used to derive PFOA references are listed in Table 3. A range of critical effects were identified by the different states. The USEPA, Massachusetts, Minnesota, Nevada, Vermont, and Wisconsin chose the reduced ossification of phalanges and accelerated puberty in mice as the critical effect. The ATSDR, Michigan, and Washington identified neurodevelopmental as well as skeletal effects in mice. New Hampshire and New Jersey listed altered liver function as the critical effect. Finally, California listed increased oxidative DNA damage and changes in mitochondrial membrane potential in liver as the specific critical effect.

Point of departures (PODs), uncertainty factors (UFs), and reference doses (RfDs) for PFOA are listed in Table 4. Using the NOAEL or LOAEL for the critical effect chosen by ATSDR or the state, a POD was identified. These PODs are expressed as the human equivalent dose (HED) for the NOAEL or LOAEL observed in the animal study and range from 0.00014 to 0.0053 mg/kg-d. Conversion of the animal dose to an equivalent human dose requires a Dosimetric Adjustment Factor (DAF), which is based upon assumptions regarding the volume of distribution and half-life of PFOA. Most states used a DAF of 1.4×10^{-4} L/kg-d, but ATSDR used 9.9×10^{-5} and New Jersey used 1.6×10^{-4}. Differences in DAF help explain, for example, how ATSDR and Michigan derive somewhat different POD HED values from the same critical effect in the same animal study.

These PODs were divided by the total UF to derive a RfD for PFOA. [Note: Technically, the ATSDR value is termed a Minimal Risk Level, or MRL). States chose a total UF ranging from 100 to 1000 (individual UFs are identified in Table 4). Reference doses for PFOA range from 0.45 to 20 ng/kg-d. Drinking water exposure assumptions for PFOA are listed in Table 5. The USEPA and Massachusetts chose the lactating woman as the receptor of concern for PFOA exposure. New Jersey and Nevada chose an adult as the receptor of concern. Wisconsin, Vermont, and the ATSDR chose a child less than a year old (infant) as the receptor of concern. The ATSDR also calculated criteria for an adult receptor. California used a lifetime average normalized drinking water intake rate. Minnesota modeled lifetime intake through breastmilk for 1 year of breast feeding followed by continuous exposure in drinking water. This model was also used by Michigan, New Hampshire, and Washington. Relative source

contributions (RSCs) for PFOA ranged from 0.2 to 1. Several states referenced the USEPA decision tree for selecting a RSC value. In some cases, the 0.2 value was based on the recommended USEPA default. In other instances, states used information on blood concentrations of PFOA in the population and a target blood concentration limit (corresponding to the RfD) to determine an RSC. The calculated drinking water limit was used as the promulgated or proposed drinking water criteria for PFOA for all states except California. They determined that the calculated value (2 ng/L; Table 5)) was below the detection limit for PFOA in water, and chose instead a detection limit of 5.1 ng/L for their criterion (Table 2).

The USEPA also considered potential carcinogenic effects of PFOA. Based upon Leydig cell testicular tumors in rats in a rodent bioassay and findings of a possible link between PFOA exposure and testicular and renal tumors in humans, the USEPA has determined that there is *Suggestive Evidence of Carcinogenic Potential* of PFOA in humans. The USEPA also noted that the International Agency for Research on Cancer has classified PFOA as *Possibly Carcinogenic in Humans*. USEPA benchmark dose modeling of Leydig cell tumor data in rats resulted in a $BMDL_{04}$ (the 95% lower confidence limit on a 4% excess probability of response) of 1.99 mg/kg-day, which corresponded to a HED of 0.58 mg/kg-d and resulted in a cancer slope factor of 0.07 $(mg/kg\text{-}d)^{-1}$. Using this cancer slope factor, the USEPA calculated drinking water concentration corresponding to a 1 E-06 excess cancer risk assuming a drinking water ingestion rate of 2.5 L/day and a default adult body weight of 80 kg. The drinking water HAL derived using the cancer slope factor was 500 ng/L, compared with 70 ng/L based upon non-cancer effects of PFOA. Because the value was higher than the non-cancer value, the latter was used as the basis for the USEPA HAL. With the exception of California, other states have explicitly or implicitly accepted the conclusion that a risk-based criterion for PFOA is driven by non-cancer effects.

Recently, California derived a cancer slope factor (CSF) for PFOA using hepatic and pancreatic tumors in male rats as the critical effect. For each tumor site, California's Office of Environmental Health Hazard Assessment (OEHHA) derived a point of departure using the linear multistage cancer model from USEPA's BMD software. The 95% lower confidence limit on the dose associated with a 5% increased risk of developing a tumor was identified as the POD. Body weight scaling to the ¾ power was used to calculate a human equivalent POD of 3.5 E-04 mg/kg-d and a cancer slope factor of 143 $(mg/kg\text{-}d)^{-1}$. Because the toxicity data suggest early-life exposures to PFOA do not significantly increase tumor formation later in life, OEHHA did not apply age sensitivity factors for the derivation of the cancer slope factor. A lifetime average drinking water rate of 0.053 L/kg-d was used to calculate a one in a million cancer risk criterion of 0.1 ng/L PFOA. As with the non-cancer criterion described above, this value is below the detection limit for PFOA determined by California, and a detection limit of 5.1 ng/L is used as their PFOA criterion (Table 2).

PFOS

Three critical effects were identified in the derivation of reference doses for PFOS (Table 6). The USEPA, ATSDR, Massachusetts, Michigan, Nevada, Vermont, and Wisconsin all listed reduced pup body weight from the Luebker et al. study as a critical effect. The ATSDR, Michigan, and Wisconsin also listed delayed eye opening from this study as a critical effect. California, Minnesota, New Hampshire, New Jersey, and Washington chose suppressed immune response in mice from Dong et al. 2009 or Dong et al. 2011. PODs, UFs, and RfDs for PFOS are listed in Table 7. PODs range from 0.0000546 to 0.000515 mg/kg-d. To obtain these HEDs, a variety of DAFs were used, reflecting different interpretation of the data regarding the toxicokinetics of PFOS. A DAF of 1.3 (or 1.28) x 10^{-4} L/kg-d was used by Minnesota, New Hampshire, and Michigan, while the USEPA, Massachusetts, and New Jersey used 8.1 or 8.2 x

10^{-5} L/kg-d. PODs were divided by a UFs ranging from 30 to 300, with ATSDR, Michigan, and Wisconsin applying an additional Modifying Factor (MF) of 10 (individual UFs and MFs are identified in Table 7). Reference doses for PFOS range from 1.8 to 20 ng/kg-d. Drinking water exposure assumptions from PFOS are listed in Table 8. Receptors of concern for PFOS in drinking water, exposure assumptions, and RSCs are identical to those chosen for PFOA. As with PFOA, RSC values range from 0.2 to 1, with some based on the USEPA default of 0.2, while others were developed based upon serum concentrations in the population intended to represent background exposure and a target serum concentration limit based upon the RfD. Minnesota and Washington used two age-dependent RSC values — 0.5 for infants and children (or young children) and 0.2 for older receptors.

The USEPA determined that there is *Suggestive Evidence of Carcinogenic Potential* for PFOS based upon liver and thyroid tumors observed in rats. However, they concluded that there was a lack of dose-response relationship for these tumors and did not develop a cancer slope factor. California recently derived a cancer slope factor for PFOS using hepatocellular adenomas in male rats and hepatocellular adenomas/carcinomas in female rats as the critical effects. OEHHA derived a POD using the linear multistage cancer model from USEPA's BMD software. The 95% lower confidence limit on the dose associated with a 5% increased risk of developing a tumor was identified as the POD. Body weight scaling to the $1/8^{th}$ power (adjustment for pharmacodynamics differences between animals) was used to calculate a human equivalent POD of 0.0011 mg/kg-d. These PODs result in cancer slope factors for PFOS of 45.5 $(mg/kg\text{-}d)^{-1}$ for males and 35.7 $(mg/kg\text{-}d)^{-1}$ for females. The higher cancer slope factor was used to drive a drinking water criterion corresponding to a 1 E-06 excess cancer risk. Because the toxicity data suggest early-life exposures to PFOS do not significantly increase tumor formation later in life, OEHHA did not apply age sensitivity factors for the derivation of the cancer slope factors. A lifetime average drinking water rate of 0.053 L/kg-d was used to calculate a one in a million cancer risk criterion of 0.4 ng/L PFOS. As with the non-cancer criterion developed by California described above, this value is below the detection limit for PFOS determined by California, and a detection limit of 6.5 ng/L is used as their PFOS criterion (Table 2).

New York Management Approach for PFOA and PFOS

New York lists criteria of 10 ng/L for PFOA and PFOS. The derivation of their drinking water criteria differed from other states. Briefly, the New York State Drinking Water Council reviewed other state and agency derivation of drinking water criteria. They identified the range of scientifically defensible criteria as 4 to 35 ppt. The Council then chose four possible drinking water criteria including the lowest value (4 ppt), 10 ppt, 20 ppt, and the highest value (35 ppt). Impacts for adopting each of the proposed criteria were discussed including number of water systems that would be out of compliance, reporting limits, and monitoring and compliance costs. Based on this discussion, the council recommended the state adopt the PFOA and PFOS criteria of 10 ppt. The state of New York accepted the council's recommendation and adopted a PFOA and PFOS criteria of 10 ppt.

Other PFAS

The ATSDR, Illinois, Massachusetts, Michigan, New Hampshire, New Jersey, Ohio, Vermont, and Washington also developed a drinking water criterion for PFNA. The critical effects identified for PFNA include reduced pup weight and developmental delays in mice and increased liver weight in pups with prenatal exposure, all from the study of Das et al. (2015) (Table 9). Some states used benchmark dose modeling to determine a threshold dose from this study, while others used a NOAEL (Table 10). The ATSDR, Michigan, New Hampshire, and New Jersey each used a different DAF for PFNA. The POD HED values ranged from 0.00043

to 0.001 mg/kg-d. Total UFs ranged from 100 to 1000. The PFNA RfDs ranged from 0.74 to 4.3 ng/kg-d PFNA. Table 11 lists the PFNA exposure assumptions. Receptors of concern include a child (0-1 year), an adult, or lifetime exposure beginning at birth. The ATSDR chose an RSC of 1, while the other states used an RSC of 0.5. Massachusetts and Vermont did not calculate a criterion for PFNA using chemical-specific data, but instead applied their PFOA and PFOS criteria (20 ng/L for both PFAS in both states) to PFNA. We were unable to locate the basis for the Ohio and Illinois PFNA criteria, so they are also absent from the comparisons in Tables 9-11.

The ATSDR, Illinois, Massachusetts, Michigan, Minnesota, New Hampshire, Ohio, Vermont and Washington developed a drinking water criterion for PFHxS. Critical effects for PFHxS include thyroid follicular cell hypertrophy and hyperplasia in rats, reduced serum thyroxine in rats, decreased litter size and reproductive toxicity in mice, and increased liver weight and centrilobular hepatocellular hypertrophy in rats (Table 12). PFHxS reference doses are summarized in Table 13. The ATSDR estimated the threshold dose for toxicity using a NOAEL while the states all used benchmark dose modeling. A variety of DAFs were used to obtain a HED: ATSDR used 6.42×10^{-5}, Minnesota and Michigan used 9.0×10^{-5}, and New Hampshire used 8.61×10^{-5} L/kg-d. PODs ranged from 0.0012 to 0.0047 mg/kg-d. Total UFs were consistently 300. Reference doses for PFHxS include 4, 9.7, and 20 ng/kg-d. Table 14 lists the PFHxS exposure assumptions. Receptors of concern include an adult, a child, and lifetime exposure beginning at birth. The ATSDR chose an RSC of 1, while the other states used an RSC of 0.5. Massachusetts and Vermont did not calculate a criterion for PFHxS using chemical-specific data, but instead applied their PFOA and PFOS criteria (20 ng/L for both PFAS in both states) to PFHxS. We were unable to locate the basis for the Ohio and Illinois PFHxS criteria, so they are also absent from the comparisons in Tables 12-14.

The states of Illinois, Massachusetts, Michigan, Minnesota, Nevada, Ohio, and Washington developed a drinking water criterion for PFBS. The USEPA developed an RfD for PFBS, but has not yet derived a HAL or other guidance value for PFBS in water. Critical effects identified included reduction in thyroid hormones in newborn offspring of mice dosed during pregnancy from the Feng et al., 2017 study. Other critical effects were taken from two studies by Lieder et al., (2009a,b) and include increased incidence of kidney hyperplasia in rats and kidney hyperplasia in parent and offspring in a 2-generational study in rats (Table 15). All of the threshold dose estimates were based upon benchmark dose modeling of toxicity data from the critical studies. PODs ranged from 0.089 to 18.9 mg/kg-d (Table 16). The POD used by Nevada comes from a USEPA PPRTV developed in 2014. In that analysis, a DAF of 0.24 was used based upon comparison of animal to human body weight. More recent analyses use a DAF derived from assumptions regarding the toxicokinetics of PFBS, which is consistent with DAFs for other PFAS used by the USEPA, ATSDR, and states. These DAFs are orders of magnitude lower and more accurately represent the difference in PFAS toxicokinetics between laboratory animals and humans. Total UFs were either 300 or 1000, and the resulting RfDs ranged from 230 to 20,000 ng/kg-d. Table 17 lists the PFBS exposure assumptions. Receptors of concern include an adult, lactating women, and lifetime exposure beginning at birth. The formula for calculating groundwater concentrations limits in Nevada does not have an RSC term, so the value is, in effect, 1. The other states used a default RSC of 0.2. We were unable to locate the basis for the Illinois, Massachusetts, or Ohio PFBS criteria, so they are absent from the comparisons in Tables 15-17.

Only one state, Michigan, was identified with a proposed drinking water limit for PFHxA. The critical effect selected by Michigan is renal tubular degeneration and renal papillary necrosis in rats. Benchmark dose modeling of the data identified a $BMDL_{10}$ of 90.4 mg/kg-d. In

the absence of adequate toxicokinetic data for PFHxA, a HED of 24.8 mg/kg-d based upon extrapolation from rats to humans using body weight. A total UF of 300 was selected (UF_H 10, UF_A 3, UF_s 1, UF_L 1, UF_D 10), yielding a RfD of 0.083 mg/kg/day (83,000 ng/kg-day). Based on an adult as the receptor, a drinking water ingestion rate of 3.353 L/d, a body weight of 80 kg, and an RSC of 0.2 were used to derive a drinking water concentration limit of 400,000 ng/L.

North Carolina, Michigan, and Ohio have drinking water criteria for GenX (Table 2). North Carolina identified the critical effect for GenX exposure as liver toxicity. A NOAEL of 0.1 mg/kg-d was used as the point of departure (POD). A total UF of 1000 was applied to the POD to derive a RfD of 1E-04 mg/kg-d, or 100 ng/kg-d. The receptor of concern is a bottle fed infant and the criterion of 140 ng/L was derived using a drinking water ingestion rate of 1.1 L/day, a body weight of 7.8 kg, and an RSC of 0.2. Michigan also identified liver toxicity as the critical effect (single cell necrosis in mice) and used benchmark dose modeling to obtain a $BMDL_{10}$ of 0.15 mg/kg-d. Using mouse and human body weight, a DAF of 0.15 was obtained, resulting in a POD HED of 0.023 mg/kg-d. A total UF of 300 (UF_H 10, UF_A 3, UF_S 3, UF_L 1, UF_D 3) was applied to the POD HED to calculate the RfD, 77 ng/kg-d. As with the PFHxA criterion, Michigan based the GenX criterion of 370 ng/L on an adult receptor, with a drinking water ingestion rate of 3.353 L/day, a body weight of 80 kg, and an RSC of 0.2. We were unable to locate the basis for the Ohio GenX criterion of 700 ng/L.

PFHpA and PFDA drinking water criteria in Table 2 for Massachusetts and PFHpA for Vermont, were not based upon specific toxicity data for these chemicals, but rather an assumption that their toxicity would be similar to PFOA and PFOS. Thus, the same criteria developed for PFOA and PFOS were used for these PFAS as well.

While nearly all states have information about PFAS on a web page, most still do not have clearly articulated drinking water criteria. Many without their own criteria mention the USEPA HALs, but it is often not apparent from information presented whether or how they are using those criteria. In preparing this report, we note that the toxicity and regulation of PFAS in drinking water is a rapidly evolving field. The information included in these tables is current as of the date of this letter, but it is reasonable to anticipate new or changing PFAS criteria from states in the near future.

Please let us know if you have any questions regarding this review.

Sincerely,



Leah D. Stuchal, Ph.D.



Stephen M. Roberts, Ph.D.

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Alaska

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California

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Delaware

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Maine

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Maryland

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Massachusetts

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Michigan

MDHHS (February 22, 2019) *Public heath drinking water screening levels for PFAS.* Michigan Department of Health and Human Services, Division of Environmental Health, Michigan PFAS Action Response Team Human Health Workgroup.

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Minnesota

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Nevada

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New Hampshire

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New Jersey

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New York

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North Carolina

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Ohio

Ohio EPA (December 2019) *Ohio Per- and Polyfluoroalkyl Substances (PFAS) Action Plan for Drinking Water.* Ohio Environmental Protection Agency and the Ohio Department of Health.

Vermont

VDOH (July 10, 2018) *Drinking Water Health Advisory for Five PFAS (per- and polyfluorinated alkyl substances).* State of Vermont, Department of Health, Agency of Human Services.

Washington

WDOH (November, 2019) *Draft Recommended State Action Levels for Per- and Polyfluoroalkyl Substances (PFAS) in Drinking Water: Approach, Methods, and Supporting Information*. Washington Department of Health, Office of Environmental Public Health Services.

Wisconsin

WDOH (June 2019) *Perfluorooctanoic acid (PFOA), 2019 Cycle 10.* Wisconsin Department of Health Services.

Table 1. States that Use the EPA HALs for PFOA and PFOS*

State	Comment
Alaska	
Colorado	
Connecticut	Drinking Water Action Level is based upon the EPA HAL expanded to include the sum of PFOA, PFOS, PFNA, PFHxS, PFHpA.
Delaware	
Florida	Florida did not adopt the EPA HALs, but developed numbers that are numerically the same using the EPA reference doses for PFOA and PFOS
Maine	Remedial Action Guidelines listed as 0.4 µg/L for PFOA and PFOS in residential water, but recommends "that the EPA health advisory level be applied at sites where groundwater is currently being used, or may be used in the future, for human consumption."
Montana	
Ohio	

* 70 ng/L for PFOA, PFOS, and PFOA + PFOS

Table 2. ATSDR and State PFAS Drinking Water Criteria Not Based Upon USEPA HALs.

	PFOA	PFOS	PFNA	PFHxS	PFHpA	PFDA	PFBA	PFHxA	PFBS	GenX
ATSDR, adult child	78 21	52 14	78 21	517 140						
California [b]	5.1	6.5								
Florida	70	70								
Illinois [c]	21	14	21	140					140,000	
Massachusetts	20 [a]	20 [a]	20 [a]	20 [a]	20 [a]	20 [a]			2000	
Michigan	8	51	9	84				400,000	1000	370
Minnesota	35	15		47			7		2000	
Nevada	667	667							667,000	
New Hampshire	12	15	11	18						
New Jersey	14	13	13							
New York [c,d]	10	10								
North Carolina										140
Ohio			21	140					140,000	700
Vermont	20 [a]	20 [a]	20 [a]	20 [a]	20 [a]					
Washington [c]	10	15	14	70					1300	
Wisconsin	20 [a]	20 [a]								

All concentrations in ng/L

[a] individually and as the sum of listed PFAS;

[b] numbers listed are management values based upon detection limit. Non-cancer risk-based values for PFOA and PFOS are 2 and 7 ng/L and cancer risk-based values are 0.1 and 0.4 ng/L, respectively.

[c] proposed

[d] management values

Table 3. Critical Effects Used to Derive PFOA Reference Doses

	Critical Effect	Study
USEPA	Reduced ossification of phalanges and accelerated puberty in mice	Lau et al. 2006
ATSDR	Neurodevelopmental and skeletal effects in mice	Koskela et al. 2016 Onishchenko et al. 2011
California	Increased oxidative DNA damage, changes in mitochondrial membrane potential, and increased biomarkers of apoptosis in the liver of female mice	Li et al., 2017
Florida*	Reduced ossification of phalanges and accelerated puberty in mice	Lau et al. 2006
Massachusetts	Reduced ossification of phalanges and accelerated puberty in mice	Lau et al. 2006
Michigan	Neurodevelopmental and skeletal effects in mice	Koskela et al. 2016 Onishchenko et al. 2011
Minnesota	Reduced ossification, accelerated puberty, trend for decreased pup body weight, increased maternal liver weight in mice	Lau et al. 2006
Nevada	Reduced ossification of phalanges and accelerated puberty in mice	Lau et al. 2006
New Hampshire	Altered liver function	Loveless et al., 2006
New Jersey	Altered liver function	Loveless et al., 2006
Vermont	Reduced ossification of phalanges and accelerated puberty in mice	Lau et al. 2006
Washington	Neurodevelopmental and skeletal effects in mice	Koskela et al. 2016 Onishchenko et al. 2011
Wisconsin	Reduced ossification of phalanges and accelerated puberty in mice	Lau et al. 2006

* Florida did not select this effect independently, but used the EPA reference dose that is based upon this critical effect.

Table 4. PFOA Reference Doses

	POD HED (mg/kg-d)	NOAEL/LOAEL	UF_H	UF_A	UF_S	UF_L	UF_D	Total UF	MF	RfD* (ng/kg-d)
USEPA	0.0053	LOAEL	10	3	1	10	1	300	--	20
ATSDR	0.000821	LOAEL	10	3	1	10	1	300	--	3
California	0.00014	LOAEL	10	3	1	3	3	300	--	0.45
Florida**	0.0053	LOAEL	10	3	1	10	1	300	--	20
Massachusetts	0.0053	LOAEL	10	3	1	10	3	1000	--	5.3
Michigan	0.001163	LOAEL	10	3	1	3	3	300	--	4
Minnesota	0.0053	LOAEL	10	3	1	3	3	300	--	18
Nevada	0.0053	LOAEL	10	3	1	10	1	300	--	20
New Hampshire	0.00061	NOAEL	10	3	1		3	100	--	6.1
New Jersey	0.00061	NOAEL	10	3	1	1	10	300	--	2
Vermont	0.0053	LOAEL	10	3	1	10	1	300	--	20
Washington	0.000821	LOAEL	10	3	1	1	1	300	--	3
Wisconsin	0.00054	LOAEL	10	3	1	10	1	300	--	2

POD = Point of Departure; HED = Human Equivalent Dose; NOAEL = no observable adverse effect level; LOAEL = lowest observable adverse effect level; UF_H = human variability uncertainty factor; UF_A = interspecies differences uncertainty factor; UF_S = duration of exposure uncertainty factor; UF_L = LOAEL to NOAEL uncertainty factor; UF_D = database uncertainty factor; MF = Modifying Factor

* ATSDR value is termed Minimal Risk Level, or MRL.

** Florida did not select these inputs independently, but used the EPA reference dose that is based upon these values.

Table 5. PFOA Exposure Assumptions

	Receptor	Ingest. Rate (L/d)	Body Wt. (kg)	Normalized Intake (L/kg-d)	RSC	Calculated Limit (ng/L)
USEPA	Lactating woman	--	--	0.054	0.2	70
ATSDR	Adult Child (0-1 yr)	3.092 1.113	80 7.8	-- --	1 1	78 21
California	Lifetime	--	--	0.053	0.2	2
Florida	Lactating woman	--	--	0.054	0.2	70
Massachusetts	Lactating woman	--	--	0.054	0.2	20
Michigan	Lifetime beginning at birth	Minnesota model			0.5	8
Minnesota	Lifetime beginning a birth	Modeled intake through breastmilk for 1 year followed by continuous direct exposure at 95th percentile rate			0.5	35
Nevada*	Adult	2.5	70	--	1	667
New Hampshire	Lifetime beginning a birth	Minnesota model			0.5	12
New Jersey	Adult	2	70	--	0.2	14
Vermont	Infant (0- 1 yr)			0.175	0.2	20
Washington	Lifetime beginning a birth	Minnesota model			0.5	10
Wisconsin	Young child	1	10	--	1	20

* Nevada tap water formula is based on 26 years of exposure at 350 days per year, and includes an inhalation component

Table 6. Critical Effects Used to Derive PFOS Reference Doses

	Critical Effect	Study
USEPA	Reduced rat pup body weight	Luebker et al., 2005
ATSDR	Delayed eye opening and decreased rat pup weight	Luebker et al., 2005
California	Decreased plaque forming cell response	Dong et al., 2009
Florida*	Reduced rat pup body weight	Luebker et al., 2005
Massachusetts	Reduced rat pup body weight	Luebker et al., 2005
Michigan	Delayed eye opening and decreased rat pup weight	Luebker et al., 2005
Minnesota	Suppressed immune response in mice	Dong et al. 2011
Nevada	Reduced rat pup body weight	Luebker et al., 2005
New Hampshire	Suppressed immune response in mice	Dong et al. 2011
New Jersey	Suppressed immune response in mice	Dong et al., 2009
Vermont	Reduced rat pup body weight	Luebker et al., 2005
Washington	Suppressed immune response in mice	Dong et al. 2011
Wisconsin	Delayed eye opening and decreased rat pup weight	Luebker et al., 2005

* Florida did not select this effect independently, but used the EPA reference dose that is based upon this critical effect.

Table 7. PFOS Reference Doses

	POD HED (mg/kg-d)	NOAEL/LOAEL	UF_H	UF_A	UF_S	UF_L	UF_D	Total UF	MF	RfD* (ng/kg-d)
USEPA	0.00051	NOAEL	10	3	1	1	1	30	--	20
ATSDR	0.000515	NOAEL	10	3	1	1	1	30	10	2
California	0.0000546	NOAEL	10	3	1	1	1	30	--	1.8
Florida**	0.00051	NOAEL	10	3	1	1	1	30	--	20
Massachusetts	0.00051	NOAEL	10	3	1	1	1	100	--	5.1
Michigan	0.0000866	NOAEL	10	3	1	1	1	30	--	2.89
Minnesota	0.000307	NOAEL	10	3	1	1	3	100	--	3.1
Nevada	0.00051	NOAEL	10	3	1	1	1	30	--	20
New Hampshire	0.000302	NOAEL	10	3	1	1	3	100	--	3.0
New Jersey	0.000055	NOAEL	10	3	1	1	1	30	--	1.8
Vermont	0.00051	NOAEL	10	3	1	10	1	30	--	20
Washington	0.000302	NOAEL	10	3	1	1	3	100	--	3.0
Wisconsin	0.000515	NOAEL	10	3	1	1	1	30	10	2

POD = Point of Departure; HED = Human Equivalent Dose; NOAEL = no observable adverse effect level; LOAEL = lowest observable adverse effect level; UF_H = human variability uncertainty factor; UF_A = interspecies differences uncertainty factor; UF_S = duration of exposure uncertainty factor; UF_L = LOAEL to NOAEL uncertainty factor; UF_D = database uncertainty factor; MF = Modifying Factor

* ATSDR value is termed Minimal Risk Level, or MRL.

** Florida did not select these inputs independently, but used the EPA reference dose that is based upon these values.

Table 8. PFOS Exposure Assumptions

	Receptor	Ingest. Rate (L/d)	Body Wt. (kg)	Normalized Intake (L/kg-d)	RSC	Calculated Limit (ng/L)
USEPA	Lactating woman	--	--	0.054	0.2	70
ATSDR	Adult Child (0-1 yr)	3.092 1.113	80 7.8	-- --	1 1	52 14
California	Lifetime	--	--	0.053	0.2	7
Florida	Lactating woman	---	--	0.054	0.2	70
Massachusetts	Lactating woman	--	--	0.054	0.2	20
Michigan	Lifetime beginning a birth	Minnesota model			0.5	51
Minnesota	Lifetime beginning a birth	Modeled intake through breastmilk for 1 year followed by continuous direct exposure at 95th percentile rate			0.5 infants and young children 0.2 older age groups	15
Nevada*	Adult	2.5	70	--	1	667
New Hampshire	Lifetime beginning a birth	Minnesota model			0.5	15
New Jersey	Adult	2	70	--	0.2	13
Vermont	Infant (0- 1 yr)			0.175	0.2	20
Washington	Lifetime beginning a birth	Minnesota model			0.5 infants and children 0.2 adults	15
Wisconsin	Young child	1	10		1	20

* Nevada tap water formula is based on 26 years of exposure at 350 days per year, and includes an inhalation component

Table 9. Critical Effects Used to Derive PFNA Reference Doses

	Critical Effect	Study
ATSDR	Reduced pup weight and developmental delays in mice.	Das et al., 2015
Florida	Not Applicable*	
Michigan	Reduced pup weight and developmental delays in mice.	Das et al., 2015
New Hampshire	Increased liver weight in pups with prenatal exposure	Das et al., 2015
New Jersey	Increased liver weight in pups with prenatal exposure	Das et al., 2015
Washington	Reduced pup weight and developmental delays in mice	Das et al., 2015

* Florida does not have a groundwater screening level for PFNA.

Table 10. PFNA Reference Doses

	POD HED (mg/kg-d)	Threshold	UF_H	UF_A	UF_S	UF_L	UF_D	Total UF	MF	RfD* (ng/kg-d)
ATSDR	0.001	NOAEL	10	3	1	1	10	300	--	3
Florida**	NA	NA	NA	NA	NA	NA	NA	NA		NA
Michigan	0.000665	NOAEL	10	3	1	1	1	300	--	2.2
New Hampshire	0.00043	$BMDL_{10}$	10	3	1	1	3	100	--	4.3
New Jersey	0.00074	$BMDL_{10}$	10	3	10	1	3	1000	--	0.74
Washington	0.001	NOAEL	10	3	1	1	10	300	--	3

POD = Point of Departure; HED = Human Equivalent Dose; NOAEL = no observable adverse effect level; LOAEL = lowest observable adverse effect level; UF_H = human variability uncertainty factor; UF_A = interspecies differences uncertainty factor; UF_S = duration of exposure uncertainty factor; UF_L = LOAEL to NOAEL uncertainty factor; UF_D = database uncertainty factor; MF = Modifying Factor; NA = Not applicable

* ATSDR value is termed Minimal Risk Level, or MRL.

* Florida does not have a groundwater screening level for PFNA.

Table 11. PFNA Exposure Assumptions

	Receptor	Ingest. Rate (L/d)	Body Wt. (kg)	Normalized Intake (L/kg-d)	RSC	Calculated Limit (ng/L)
ATSDR	Adult Child (0-1 yr)	3.092 1.113	80 7.8	-- --	1 1	78 21
Florida*	NA	NA			NA	NA
Michigan	Lifetime beginning at birth	Minnesota model			0.5	9
New Hampshire	Lifetime beginning a birth	Minnesota model			0.5	11
New Jersey	Adult	2	70	--	0.5	13
Washington	Lifetime beginning a birth	Minnesota model			0.5	14

NA = Not applicable

* Florida does not have a groundwater screening level for PFNA.

Table 12. Critical Effects Used to Derive PFHxS Reference Doses

	Critical Effect	Study
ATSDR	Thyroid follicular cell hypertrophy and hyperplasia in rats	Butenhoff et al. 2009; Hoberman and York, 2003
Florida	Not applicable*	
Michigan	Thyroid follicular cell hypertrophy and hyperplasia, and increased liver weight and centrilobular hepatocellular hypertrophy in rats	Butenhoff et al. 2009; Hoberman and York, 2003
Minnesota	Reduced serum thyroxine in rats	NTP, 2018
New Hampshire	Decreased litter size and reproductive toxicity in mice	Chang et al., 2018
Washington	Reduced serum thyroxine in rats	NTP, 2018

* Florida does not have a groundwater screening level for PFHxS.

Table 13. PFHxS Reference Doses

	POD HED (mg/kg-d)	Threshold	UF_H	UF_A	UF_S	UF_L	UF_D	Total UF	MF	RfD* (ng/kg-d)
ATSDR	0.0047	NOAEL	10	3	1	1	10	300	--	20
Florida**	NA	NA	NA	NA	NA	NA	NA	NA		NA
Michigan	0.00292	$BMDL_{20}$	10	3	1	1	1	300	--	9.7
Minnesota	0.00292	$BMDL_{20}$	10	3	1	1	10	300	--	9.7
New Hampshire	0.0012	BMDL	10	3	3	1	3	300	--	4
Washington	0.00292	$BMDL_{20}$	10	3	1	1	10	300	--	9.7

POD = Point of Departure; HED = Human Equivalent Dose; NOAEL = no observable adverse effect level; LOAEL = lowest observable adverse effect level; UF_H = human variability uncertainty factor; UF_A = interspecies differences uncertainty factor; UF_S = duration of exposure uncertainty factor; UF_L = LOAEL to NOAEL uncertainty factor; UF_D = database uncertainty factor; MF = Modifying Factor; NA = not applicable

* ATSDR value is termed Minimal Risk Level, or MRL.

** Florida does not have a groundwater screening level for PFHxS.

Table 14. PFHxS Exposure Assumptions

	Receptor	Ingest. Rate (L/d)	Body Wt. (kg)	Normalized Intake (L/kg-d)	RSC	Calculated Limit (ng/L)
ATSDR	Adult Child (0-1 yr)	3.092 1.113	80 7.8	-- --	1 1	517 140
Florida*	NA	NA			NA	NA
Michigan	Lifetime beginning a birth	Minnesota model			0.5	84
Minnesota	Lifetime beginning a birth	Modeled intake through breastmilk for 1 year followed by continuous direct exposure at 95th percentile rate			0.5	47
New Hampshire	Lifetime beginning a birth	Minnesota model			0.5	18
Washington	Lifetime beginning a birth	Minnesota model			0.5	70

NA = Not applicable

* Florida does not have a groundwater screening level for PFHxS.

Table 15. Critical Effects Used to Derive PFBS Reference Doses

	Critical Effect	Study
EPA	Reduction in thyroid hormones in newborn offspring of mice dosed during pregnancy.	Feng et al., 2017
Florida	Not applicable*	
Michigan	Increased incidence of kidney hyperplasia in rats.	Lieder et al., 2009b
Minnesota	Hyperplasia in kidney in parent and offspring in 2-gen study in rats	Lieder et al., 2009a
Nevada	Increased incidence of kidney hyperplasia	Lieder et al., 2009b
Washington	Reduction in thyroid hormones in newborn offspring of mice dosed during pregnancy.	Feng et al., 2017

* Florida does not have a groundwater screening level for PFBS.

Table 16. PFBS Reference Doses

	POD HED (mg/kg-d)	Threshold	UF_H	UF_A	UF_S	UF_L	UF_D	Total UF	MF	RfD (ng/kg-d)
EPA	4.2	$BMDL_{20}$	10	3	1	1	10	300	--	10,000
Florida*	NA	NA	NA	NA	NA	NA	NA	NA		NA
Michigan	0.225	$BMDL_{10}$	10	3	10	1	3	1000	--	230
Minnesota	0.129	$BMDL_{10}$	10	3	3	1	3	300	--	430
Nevada**	18.9	$BMDL_{10}$	10	3	10	1	3	1000	--	20,000
Washington	0.089	$BMDL_{20}$	10	3	1	1	10	300	--	300

POD = Point of Departure; HED = Human Equivalent Dose; NOAEL = no observable adverse effect level; LOAEL = lowest observable adverse effect level; UF_H = human variability uncertainty factor; UF_A = interspecies differences uncertainty factor; UF_S = duration of exposure uncertainty factor; UF_L = LOAEL to NOAEL uncertainty factor; UF_D = database uncertainty factor; MF = Modifying Factor; NA = not applicable

* Florida does not have a groundwater screening level for PFBS.

** based upon EPA PPRTV value from 2014.

Table 17. PFBS Exposure Assumptions

	Receptor	Ingest. Rate (L/d)	Body Wt. (kg)	Normalized Intake (L/kg-d)	RSC	Calculated Limit (ng/L)
Florida*	NA	NA	NA	NA		NA
Michigan	Lifetime beginning at infancy	--	--	0.044	0.2	1000
Minnesota	Lifetime beginning at infancy	--	--	0.044	0.2	2000
Nevada	Adult	2.5	70	--	1	667,000
Washington	Lactating women	--	--	0.047	0.2	1300

NA = Not applicable

* Florida does not have a groundwater screening level for PFBS.