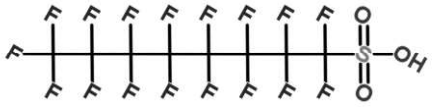
Fact Sheet Date: April 8, 2021

NEW YORK STATE
HUMAN HEALTH FACT SHEET

Ambient Water Quality Value for
Protection of Human Health and Sources of Potable Water[1]

SUBSTANCE: Perfluorooctane Sulfonic Acid (PFOS)



CAS REGISTRY NUMBER: 1763-23-1

AMBIENT WATER QUALITY VALUE: 0.0027 mcg/L

BASIS: Oncogenic Effects (6 NYCRR 702.4)

INTRODUCTION

Perfluorooctane sulfonic acid (perfluorooctane sulfonate, PFOS) is an environmentally persistent anthropogenic chemical that had many uses such as in fire-fighting foams and fabric stain-resistance treatments. PFOS is no longer manufactured in the United States but can be imported and used for specific limited uses. PFOS is released into the environment from fluoropolymer manufacturing or processing facilities, effluent releases from wastewater treatment plants, landfill leachates, the spreading of biosolids, and the use of aqueous fire-fighting foams (ATSDR, 2018; HC, 2018).

The toxicity of PFOS has been reviewed and summarized by several authoritative bodies (ATSDR, 2018; EFSA CONTAM, 2018; HC, 2018; NTP, 2016; NJ DEP, 2019; OECD, 2002; US EPA 2009, 2016a). These reviews identify important studies on the health effects associated with exposure to PFOS, including studies (when available) on the chronic (oncogenic and nononcogenic), developmental, and reproductive effects observed in humans and animals. We derived the ambient water quality value of 0.0027 mcg/L for PFOS using available toxicological data and risk assessments, the definitions in 6 NYCRR 700.1, and the procedures outlined in 6 NYCRR 702.2 through 702.7.

[1] A list of commonly used abbreviations and acronyms is attached as Exhibit 4.

702.3. PROCEDURES FOR DERIVING STANDARDS AND GUIDANCE VALUES BASED ON SPECIFIC MCLS AND PRINCIPAL ORGANIC CONTAMINANT CLASSES

PFOS has a Specific MCL of 0.01 mcg/L as defined in 6 NYCRR 700.1. Thus, the potential ambient water quality value for PFOS under 6 NYCRR 702.3 is 0.01 mcg/L.

702.4. PROCEDURES FOR DERIVING STANDARDS AND GUIDANCE VALUES BASED ON ONCOGENIC EFFECTS

Epidemiological studies of workers or the general population have not provided convincing evidence of increased cancer risk from PFOS exposure (ATSDR, 2018; EFSA CONTAM, 2018; US EPA, 2016a). The results of one study in occupationally exposed workers showed an association between PFOS exposure and increased incidence of bladder cancer; however, the results were considered inconclusive due to the limited size of the study cohort (Alexander and Olsen, 2007; CA EPA, 2010; EFSA CONTAM, 2018; OECD, 2002; US EPA 2016a).

There is only one study that evaluates the oncogenicity of PFOS in animals (Butenhoff et al., 2012a; OECD, 2002).[2] In this study, male and female rats were fed diets containing PFOS at concentrations of 0.5, 2, 5, or 20 parts per million (ppm) for 104 weeks.[3] A recovery group was fed diets containing 20 ppm for 52 weeks and was observed until death. PFOS increased the incidence of hepatocellular adenoma/carcinoma in male and female rats at the highest dose (20 ppm), equivalent to 0.984 milligrams per kilogram per day (mg/kg-day) in males and 1.25 mg/kg-day in females. A statistically significant increase in thyroid tumors in male rats in the recovery group was reported at the highest dose tested (0.984 mg/kg-day).[4] PFOS also increased the incidence of mammary tumors in female rats without a clear dose-response effect (Butenhoff et al., 2012a; OECD, 2002)[5]. Based on the results of this study, some agencies consider PFOS to be oncogenic in animals (EFSA, 2008; HC, 2018; NJ DEP, 2019; OECD, 2002).

[2] This study was conducted by the 3M Company in 2002 and was made publically available via a report by Thomford (2002) prior to publication in Butenhoff et al. (2012a).

[3] These dietary concentrations correspond to oral doses of 0, 0.024, 0.098, 0.242, and 0.984 mg/kg-day in males and 0, 0.029, 0.120, 0.299, and 1.25 mg/kg-day in females.

[4] The authors stated that the "observation of a statistically significant increased incidence of thyroid follicular cell adenoma in the 20 ppm recovery group males without observation of similar increases in males and/or females of the 20 ppm group is paradoxical and may represent a chance occurrence."

[5] Females had a statistically significant increase in follicular cell adenoma/carcinoma, but only at the 5-ppm dose level.

In determining whether PFOS has oncogenic effects under 6 NYCRR 700.1, we also considered oncogenicity data for a structurally similar compound, perfluorooctanoic acid (PFOA). PFOS and PFOA share similar physical and chemical properties (ATSDR, 2018; US EPA, 2016a) and are frequently found together in the environment (Kannan et al., 2005). Studies show that PFOS and PFOA are readily absorbed after oral exposure, are not metabolized in the body, and accumulate primarily in the serum, kidney, and liver. In addition, both compounds have long serum half-lives in humans, generally ranging from about 2 to 4 years for PFOA and about 4 to 6 years for PFOS (ATSDR, 2018; Olsen et al., 2007; US EPA, 2016a). PFOA and PFOS are found in humans bound to blood serum albumin (Salvalaglio et al., 2010). PFOA (Butenhoff et al., 2012b) and PFOS (Butenhoff et al., 2012a) caused liver adenomas and carcinomas in dietary studies in rodents. PFOA induces tumors at multiple sites in rats (i.e., liver, mammary gland, testicular Leydig cell, and pancreatic acinar cell tumors) and has oncogenic effects under 6 NYCRR 700.1(a)(39)(vi), based on induction of tumors in one mammalian species, reported in two independent studies (NYS, 2019). Thus, PFOS has oncogenic effects as defined under 6 NYCRR 700.1 because it induces tumors in "one mammalian species, supported by positive results for another substance for which similar oncogenic effects are anticipated because of similarity of functional groups or metabolic or toxicologic pathways."

Most of the evidence from short-term *in vitro* assays suggest that PFOS is not active in short-term tests indicative of oncogenic potential (ATSDR, 2018; EFSA, 2008; HC, 2018; OECD, 2002; US EPA, 2016a). However, some studies have shown limited positive evidence of PFOS having direct interaction with DNA, such as adduct formation in calf thymus DNA (Lu et al., 2012) as well as DNA damage (comet assay) and micronucleus formation in rat bone marrow (Celik et al., 2013).

It has been hypothesized that the tumors observed after dietary exposure of rats to PFOS may be due to activation of nuclear peroxisomal proliferator activated receptors (PPAR)[6] and other nuclear receptors (Butenhoff et al., 2012a; Jacquet et al., 2012). However, it has also been suggested that other, PPAR-independent mechanisms may be involved in PFOS carcinogenesis (EFSA CONTAM, 2018). Since the oncogenic MOA for PFOS is unknown[7], under 6 NYCRR 702.4, "the standard or guidance value shall be based

[6] PPARα regulates lipid homeostasis by altering the expression of genes involved in uptake, activation, and oxidation of fatty acids (Butenhoff et al., 2012a; Elcombe et al., 2012).

[7] US EPA (2005a) guidance recommends the use of age dependent adjustment factors (ADAFs) when assessing the cancer risks of chemicals that act through a mutagenic mode of action (MOA) for carcinogenicity. Given that the oncogenic MOA for PFOS is unknown, and the available data do not suggest that PFOS acts through a mutagenic MOA, ADAFs were not used in the derivation of potential ambient water quality values for PFOS (oncogenic effects).

on the 95 percent lower confidence limit on the human dose corresponding to an excess lifetime cancer risk of one-in-one million."

The New Jersey Department of Environmental Protection (NJ DWQI, 2018; NJ DEP, 2019)[8] evaluated the available scientific literature on the oncogenic effects of PFOS and derived a CPF for PFOS[9] based on the dose-response data for liver tumors in rats (Tables 1 and 3) reported in Butenhoff et al. (2012a). The NJ DEP used area under the curve calculations to obtain a time weighted average PFOS serum concentration for each administered dose (including the recovery group), and then modeled a serum $BMDL_{10}$ in female rats (137 mg/L), which was used as the POD.[10] Linear extrapolation from the POD yielded a rat CPF (expressed as the risk per unit of serum concentration) of 0.00073 $(mg/L)^{-1}$. The NJ DEP obtained the corresponding human cancer potency factor (9.0 $(mg/kg\text{-}day)^{-1}$) for PFOS using the same human one-compartment model the US EPA used to derive a PFOS reference dose (2016a).[11]

We derived a potential ambient water quality value (oncogenic effects) for PFOS based on the dose-response data for liver tumors in rats reported in Butenhoff et al. (2012a) using the time-weighted average (area under the curve) PFOS serum concentrations reported in NJ DEP (2019).[12] We did not include recovery groups in the dose-response modeling because the duration of exposure differed between animals in the recovery group and animals in the other dose groups. Animals in the recovery groups were exposed to PFOS via the diet for 52 weeks and were given a control diet (without PFOS) for the remainder of the 104 week study. Whereas, animals in the other dose groups were exposed to PFOS for the entire duration of the study. Based on the range of observation for liver tumor incidence reported in the Butenhoff et al. (2012a) study, we selected a BMR of

[8] The cancer potency estimate and reference dose derived by NJ DEP (2019) is also documented in an earlier report from the NJ Drinking Water Quality Institute (i.e., NJ DWQI, 2018).

[9] No other cancer potency factors for PFOS derived by authoritative bodies were located. Health Canada (2018) evaluated the oncogenic effects of PFOS and derived a tolerable daily intake (i.e., reference dose) based on the increased incidence of hepatocellular tumors in male rats. Health Canada stated that "Although the mode of action for PFOS-induced tumours has not yet been elucidated, the weight of evidence more strongly suggests that PFOS is a non-mutagenic compound. For this reason, a non-linear low-dose extrapolation approach (i.e., the tolerable daily intake (TDI) approach) is the most appropriate method for deriving a health-based value (HBV) for cancer." However, under 6 NYCRR 702.4, if "data on mode-of-action are unavailable, or if the mode-of-action analysis provides evidence of linearity at low doses or does not provide unequivocal evidence of nonlinearity at low doses, the standard or guidance value shall be based on the 95 percent lower confidence limit on the human dose corresponding to an excess lifetime cancer risk of one-in-one million." Therefore, Health Canada's tolerable daily intake was not further considered as a potential basis for an ambient water quality value for PFOS based on oncogenic effects.

[10] A $BMDL_{10}$ is the 95% LCL on the benchmark serum level (internal dose) associated with a 10% increase in liver tumors.

[11] Cancer potency factor = Risk per unit serum level / Clearance = 0.00073 $(mg/L)^{-1}$ / 0.000081 L/kg-day = 9.0 $(mg/kg/day)^{-1}$. PFOS clearance (US EPA, 2016a) = (ln2/PFOS half-life) x volume of distribution = (0.693/1971 days) x 0.23 L/kg = 0.000081 L/kg-day.

[12] Serum PFOS data were obtained from Tables 45 and 46 of NJ DEP (2019).

5% for dose-response modeling and chose the serum $BMDL_{05}$ as the POD[13], which is consistent with 6 NYCRR 702.4 and US EPA (2012a) guidance. We obtained serum $BMDL_{05}$ estimates based on liver tumors in male rats and female rats using the cancer multistage model (Tables 1 and 2). We did not consider alternate models because the multistage model adequately described the dose-response data within the range of observation (Table 2).[14] This is consistent with 6 NYCRR 702.4 and recent U.S. Environmental Protection Agency's cancer risk-assessment guidance and practice giving preference (among models that adequately described the data) to the multistage model when modeling cancer bioassay data (Gehlhaus et al., 2011; US EPA, 2005b, 2012a,b).[15]

Experimental evidence to indicate that one sex is a better surrogate for humans was not found, and our serum $BMDL_{05}$ estimates (i.e., 33,761 mcg/L for males and 62,453 mcg/L for females) differed by only about 2-fold. Thus, we selected the median serum $BMDL_{05}$ (48,107 mcg/L) as the POD and the basis of a potential ambient water quality value (oncogenic effects) for PFOS.

Using procedures consistent with those outlined in 6 NYCRR 702.4, we calculated the HED at the median serum $BMDL_{05}$ (48,107 mcg/L) using a human single-compartment model to obtain a pharmacokinetic adjustment factor (NJ DEP, 2019; US EPA, 2016a) that accounts for the large interspecies differences in PFOS serum half-lives observed in studies of humans and animals.

[13] A $BMDL_{05}$ is the 95% LCL on the benchmark (internal) dose associated with a 5% increase (relative to controls) of an effect. A BMDL is also known as an LED, which is the 95 percent lower confidence limit on the effective dose as described in 6 NYCRR 702.4.

[14] Dose-response curves were also visually inspected to ensure that the model adequately describes the data.

[15] The US EPA (2012a) noted, "in the absence of a biologically based model, dose-response modeling is largely a curve-fitting exercise among the variety of available empirical models. Currently there is no recommended hierarchy of models that would expedite model selection, in part because of the many different types of datasets and study designs affecting dose-response patterns. As more flexible models are developed, hierarchies for some categories of endpoints will likely be more feasible. Some model hierarchies could be established as preferred practices. For example, it is a current practice of US EPA's IRIS program to prefer the multistage model for cancer dose-response modeling of cancer bioassay data (Gehlhaus et al., 2011). The multistage model (in fact a family of different stage polynomial models) is sufficiently flexible for most cancer bioassay data, and its use provides consistency across cancer dose-response analyses." More specifically, to support using only the multistage model to determine the carcinogenic potency of tetrachloroethene, US EPA (2012b) noted, "The multistage model has been used by EPA in the majority of quantitative cancer assessments, initially because of its parallelism to the multistage carcinogenic process. A benefit of the multistage model is its flexibility in fitting a broad array of dose-response patterns, including allowing linearity at low dose."

$$HED_{BMDL05} = \text{serum } BMDL_{05} \times PKAF \times PDAF$$

where,

$$\text{median serum } BMDL_{05} = 48{,}107 \text{ mcg/L}$$

$$PKAF = \text{Pharmacokinetic Adjustment Factor} = 8.1 \times 10^{-5} \text{ L/kg-day*}$$

$$PDAF = \text{Pharmacodynamic Adjustment Factor} = 1\text{**}$$

$$HED_{BMDL05} = 48{,}107 \text{ mcg/L} \times 8.1 \times 10^{-5} \text{ L/kg-day} \times 1$$

$$HED_{BMDL05} = 3.9 \text{ mcg/kg-day (or } 3.9 \times 10^{-3} \text{ mg/kg-day)}$$

*$PKAF = CL_{human}$
where,
CL_{human} = Volume of Distribution x (ln 2 ÷ half-life), assuming first-order kinetics.
CL_{human} = 0.23 L/kg x (0.693 ÷ 1971 days) = 0.000081 L/kg-day (US EPA, 2016a)

**Based on evidence and analysis in US EPA (1992), we assumed that in the absence of evidence to the contrary, animals and humans are at equal lifetime excess cancer risk at equal lifetime internal doses. Therefore, the adjustment factor for pharmacodynamic differences is one.

We divided the HED_{BMDL05} by 50,000 to obtain the human risk-specific dose corresponding to the 95% LCL on the dose (7.8×10^{-5} mcg/kg-day) associated with an excess lifetime oncogenic cancer risk of one-in-one-million.[16] We selected this dose for use in the derivation of a potential ambient water quality value (oncogenic effects) for PFOS.

$$\text{Human risk-specific } 1 \times 10^{-6} \text{ Dose} = HED_{BMDL05} / 50{,}000$$

$$\text{Human risk-specific } 1 \times 10^{-6} \text{ Dose} = 3.9 \text{ mcg/kg-day} / 50{,}000$$

$$\text{Human risk-specific } 1 \times 10^{-6} \text{ Dose} = 7.8 \times 10^{-5} \text{ mcg/kg-day}$$

Using procedures that are consistent with 6 NYCRR 702.2 and 702.4, we calculated the PFOS water concentration (0.0027 mcg/L, two significant figures) corresponding to an excess lifetime cancer risk of one-in-one million using the risk-specific (1×10^{-6}) human dose (7.8×10^{-5} mcg/kg-day) assuming a 70-kg adult

[16] A dose at any lifetime excess cancer risk can be obtained from the straight line that extrapolates 5% excess lifetime cancer risk at the HED_{BMDL05} to zero excess risk at zero dose. For example, a one-in-one-million excess lifetime risk (equal to 0.000001) is 50,000-fold lower than an excess lifetime risk of 5% (equal to 0.05). Therefore, the dose at a one-in-one-million excess lifetime risk is obtained by dividing the dose at a 5% excess risk by 50,000 (equal to 0.05/0.000001).

consumes 2 liters of water per day. We selected 0.0027 mcg/L as the potential ambient water quality guidance value (oncogenic effects) for PFOS.

$$\text{Risk-Specific } (1 \times 10^{-6}) \text{ Water Concentration} = \frac{\text{Risk Specific } (1 \times 10^{-6}) \text{Dose} \times \text{Body Weight}}{\text{Drinking Water Consumption Rate}}$$

$$1 \times 10^{-6} \text{ Water Concentration} = \frac{7.8 \times 10^{-5} \text{ mcg/kg-day} \times 70 \text{ kg}}{2\text{L/day}}$$

$$1 \times 10^{-6} \text{ Water Concentration} = 0.0027 \text{ mcg/L}$$

702.5. PROCEDURES FOR DERIVING STANDARDS AND GUIDANCE VALUES BASED ON NONONCOGENIC EFFECTS

Human studies on PFOS have suggested possible links between exposure to PFOS and effects on immune response, cholesterol, birth weight, and various thyroid parameters (ATSDR, 2018; EFSA CONTAM, 2018; NTP, 2016; US EPA, 2016a). These studies are inadequate for use in dose-response assessment, due to lack of reliable quantitative exposure data (US EPA, 2016a).

The US EPA (2016a), the Minnesota Department of Health (MDH, 2019), and the NJ DEP (NJ DWQI, 2018; NJ DEP, 2019) evaluated the available animal and human studies on the nononcogenic effects of PFOS, and derived RfDs and health based-values for PFOS in drinking water based on effects observed in animals (Table 4).

The US EPA (2016a) based its RfD on developmental toxicity (reduced pup body weight) in the offspring of rats exposed to PFOS for 84 days across two generations (see Exhibit 1). The US EPA converted the NOEL of 0.1 mg/kg-day to a serum PFOS level of 6.26 mg/L using the rodent pharmacokinetic model of Wambaugh et al. (2013), and then applied a human single compartment model to obtain the corresponding human POD (i.e., an HED_{NOEL} of 0.00051 mg/kg-day).[17] Application of a total uncertainty factor of 30 (10X for intraspecies differences and 3X for interspecies pharmacodynamic differences) yielded the RfD of 2.0×10^{-5} mg/kg-day.

[17] Human equivalent dose (HED_{NOEL}) = PFOS serum concentration x PFOS clearance = 6.26 mg/L x 0.000081 L/kg-day = 0.00051 mg/kg-day. Where, PFOS clearance = (ln2/PFOS half-life) x volume of distribution = (0.693/1971 days) x 0.23 L/kg = 0.000081 L/kg-day

The MDH (2019) based its RfD on immune effects (increased interleukin 4 and decreased sheep red blood cell-specific IgM levels) in adult male mice exposed to PFOS for 60 days (see Exhibit 2). The MDH converted the measured serum PFOS level of 2.36 mg/L (corresponding to the administered dose NOEL of 0.0167 mg/kg-day) to obtain the human point of departure (an HED_{NOEL} = 0.000307 mg/kg-day)[18] using a single-compartment model based on a human clearance calculated with a shorter assumed mean half-life than was used by the US EPA (3.4 years [Li et al., 2018] compared to 5.4 years [Olsen et al., 2007]). Application of a total UF of 100 (10 for intraspecies differences, 3 for interspecies pharmacodynamic differences, and 3 for database uncertainty) yielded an RfD of 3.1 x 10^{-6} mg/kg-day.

The NJ DEP (NJ DWQI, 2018; NJ DEP, 2019) derived an RfD (2 x 10^{-6} mg/kg-day) based on immune effects (decreased plaque forming cell response) in adult male mice exposed to PFOS for 60 days (see Exhibit 3). In this study, the NOEL for immune effects is 0.0083 mg/kg-day, corresponding to a measured serum PFOS level of 0.674 mg/L, and the LOEL for these effects is 0.083 mg/kg-day (which corresponds to a PFOS serum concentration of 7.132 mg/L). The NJ DEP used the measured PFOS serum level at the NOEL as the point of departure and applied a UF of 30 (10X for intraspecies differences and 3X for interspecies pharmacodynamic differences) to obtain a target human serum level of 0.0225 mg/L. The NJ DEP calculated the RfD from the target human serum level using the same human single-compartment model used by the US EPA (2016a).[19]

Using procedures consistent with 6 NYCRR 702.5, we selected the POD used by the NJ DEP (NJ DWQI, 2018; NJ DEP, 2019) as the basis of a potential ambient water quality value (nononcogenic effects) for PFOS. The primary considerations for selecting this POD were:

- The LOEL for immune effects in the study selected by the NJ DEP (0.083 mg/kg-day) is lower than the LOEL for developmental toxicity (0.4 mg/kg-day) in the study used by the US EPA.
- Immunotoxicity is a well-established and sensitive endpoint for PFOS in animals. In addition, epidemiological studies have reported associations between serum PFOS levels and immunotoxicity (Grandjean et al., 2012; Granum, 2013; Stein et al., 2016).

[18] Human equivalent dose (HED_{NOEL}) = PFOS serum concentration x PFOS clearance = 2.36 mg/L x 0.00013 L/kg-day = 0.000307 mg/kg-day. Where, PFOS clearance = (ln2/PFOS half-life) x volume of distribution = (0.693/1241 days) x 0.23 L/kg = 0.00013 L/kg-day

[19] Clearance factor is from US EPA (2016a). RfD = PFOS target human serum level x PFOS clearance = 0.0225 mg/L x 0.000081 L/kg-day = 2 x 10^{-6} mg/kg-day.

- A recent major report on PFOS immunotoxicity by the National Toxicology Program (2016) which evaluated animal, human and *in vitro*/mechanistic studies concluded that PFOS is presumed to be an immune hazard to humans.

The NJ DEP derived their RfD using a measured PFOS serum level at the NOEL of 0.674 mg/L as the rat POD. Consistent with 6 NYCRR 702.5, this POD is expressed as a HED of 0.000055 mg/kg-day by applying the human single-compartment model used by the US EPA (2016a) to the measured PFOS serum level.[20] The total UF of 30 applied by the NJ DEP is consistent with 6 NYCRR 702.5 given the areas of uncertainty and variation.

$$RfD = HED_{NOEL} \ / \ UF$$

where,

UF = 30 (3X for interspecies differences in pharmacodynamics, 10X for inter-human variability)

$$RfD = 0.000055 \text{ mg/kg-day} / 30$$

$$RfD = 1.8 \times 10^{-6} \text{ mg/kg-day or } 0.0018 \text{ mcg/kg-day}$$

We applied the procedure outlined in 6 NYCRR 702.2 and 702.5 to derive a potential ambient water quality value for nononcogenic effects (0.013 mcg/L, rounded to two significant figures) using the selected RfD (0.0018 mcg/kg-day), a 20% relative source contribution (0.2), and assuming an adult body weight of 70 kg and a drinking-water consumption rate of 2 L/day.

$$\text{Potential Ambient Water Quality Value} = \frac{0.0018 \text{ mcg/kg-day} \times 70 \text{ kg} \times 0.2}{2 \text{ L/kg-day}} = 0.013 \text{ mcg/L}$$

The use of age-specific drinking-water consumption rates in the derivation to address the potential for children to be more sensitive than adults to the nononcogenic effects of PFOS was considered, but was not used because the weight of scientific evidence is insufficient to conclude that exposure to PFOS during childhood poses a greater risk of nononcogenic effects than exposure during adulthood (ATSDR, 2018, OECD, 2002). In addition, for the toxicological endpoint on which the ambient water quality value (nononcogenic effects) is based (immune toxicity), effects were observed at a lower PFOS exposure level in adult animals (Dong et al.,

[20] Human equivalent dose (HED_{NOEL}) = PFOS serum concentration x PFOS serum clearance = 0.674 mg/L x 0.000081 L/kg-day = 0.000055 mg/kg-day.

2009) than the maternal exposure that caused effects in young animals exposed gestationally (Luebker et al., 2005).

702.7. PROCEDURE FOR DERIVING STANDARDS AND GUIDANCE VALUES BASED ON CHEMICAL CORRELATION

Chemical-specific toxicological data are sufficient to derive potential ambient water quality values for PFOS based on both its oncogenic (6 NYCRR 702.4) and nononcogenic effects (6 NYCRR 702.5). Thus, values based on oncogenic or nononcogenic effects using chemical correlation are unnecessary.

SELECTION OF VALUE

According to 6 NYCRR 702.2(b), the ambient water quality value [Health (Water Source)] shall be the most stringent of the potential values derived using the procedures found in 6 NYCRR 702.3 through 702.7. Using procedures from 6 NYCRR 702.4 and 702.5, respectively, we derived potential ambient water quality values of 0.0027 mcg/L (oncogenic effects) and 0.013 mcg/L (nononcogenic effects) for PFOS. The most stringent of the potential values is 0.0027 mcg/L (6 NYCRR 702.4, Oncogenic Effects) and thus, this value is selected as the ambient water quality value [Health (Water Source)] for PFOS.

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SEARCH STRATEGY

We reviewed publications by various state, federal, or international public health agencies (listed in fact sheet references) and identified important papers from the list of references within each document. Before and on April 10, 2019, we also searched the biomedical literature using PubMed (U.S. National Library of Medicine) and the search term “PFOS and toxicity”.

Bureau of Toxic Substance Assessment
New York State Department of Health
August 2019

EXHIBITS

Exhibit 1. US EPA (2016a,b) Reference Dose Derivation and Lifetime Drinking Water Health Advisory for Perfluorooctane Sulfonate (PFOS).
Exhibit 2. MDH (2019) Derivation of Reference Dose and Health-based Water Value for Perfluorooctane Sulfonate.
Exhibit 3. NJ DWQI (2018) Health-based MCL for Perfluorooctane Sulfonate.
Exhibit 4. List of Abbreviations and Acronyms Frequently Used in New York State Human Health Fact Sheets.

Table 1. Exposure Response Data for Liver Tumors in Male and Female Rats.[A]

Tumor Site	Tumor Type	PFOS Time-weighted Average Serum Concentrations (mcg/L) and Tumor Incidence				
rat (male)		**25**	**2,554**	**11,724**	**31,225**	**116,950**
liver	hepatocellular adenoma	0/60	3/50	3/50	1/50	7/60[B]
rat (female)		**816**	**5,309**	**22,153**	**64,073**	**207,633**
liver	hepatocellular, adenoma/carcinoma combined	0/60	1/50	1/49	1/50	6/60[B]

[A]Tumor incidence data come from Tables 5 and 6 of the Butenhoff et al. (2012a) study. PFOS serum concentrations for this study are from Tables 45 and 46 of NJ DEP (2019) and are based on the area under the curve serum levels for each dose-group, time-weighted across the duration of the study. NJ DEP (2019) reported serum concentrations in units of ng/mL (which is equivalent to units in mcg/L).
[B]Statistically significant ($p \leq 0.05$) compared to controls.

Table 2. Results of Benchmark Dose Modeling[A] of Tumor Incidence Data from Butenhoff et al. (2012a).

Species/ Gender	Tumor Site	BMD_{05}[B] (mcg/L)	$BMDL_{05}$[C] (mcg/L)	Chi-Squared p-Value for Goodness-of-Fit[D]
rat (male)	liver	89,108	33,761	0.1873
rat (female)	liver	134,128	62,453	0.5186

[A]Benchmark Dose Software Version 3.4 (US EPA, 2012c); the multistage model is preferred for dose-response modeling of cancer bioassay data (US EPA, 2012a,b); the multistage model was run on default settings (i.e., default parameters including a 2^o polynomial).
[B]The BMD_{05} is the internal dose (PFOS serum concentration) associated with a 5% increase in tumor incidence relative to background (control) incidence.
[C]The $BMDL_{05}$ is the 95% LCL on the internal dose (PFOS serum concentration) associated with a 5% increase in tumor incidence relative to background (control) incidence.
[D]The p-value for the Chi-Squared test should be greater than 0.05 given an *a priori* selection of a model (i.e., the cancer multistage) (US EPA, 2012a), which indicates that there is no significant difference between expected (i.e., model predicted) and observed tumor incidences.

Table 3. Authoritative Body Cancer Potency Estimates for PFOS.[1]

Agency	Risk-Specific Dose[2] (mg/kg-day)	Cancer Potency Factor $(mg/kg\text{-}day)^{-1}$	Extrapolation Methods		Summary
			High to Low Dose	Animal to Human	
NYS (derived under 6 NYCRR 702.4)	7.8×10^{-8}	12.8	linearized multistage model with linear extrapolation from the POD	single-compartment human PBPK model	Based on increased incidence of hepatocellular adenomas and carcinomas in male and female rats exposed to PFOS via the diet for two years
NJ DWQI (2018)	1.1×10^{-7}	9.0	dose-response models with linear extrapolation from the POD	single-compartment human PBPK model	Based on the combined incidence of hepatocellular adenomas and carcinomas in female rats exposed to PFOS via the diet for two years.
Health Canada (2018)	--	--	uncertainty factors	chemical-specific UF of 10 (pharmaco-kinetics)[3]	Based on hepatocellular tumors in male rats exposed via the diet for two-years. Using a noncancer threshold approach, Health Canada calculated a TDI of 0.0011 mg/kg-day for carcinogenicity based on weight of evidence that suggests that PFOS is a non-mutagenic compound. The TDI is based on a $BMDL_{10}$ of 0.28 mg/kg-day and a total UF of 25 (2.5 for interspecies pharmacodynamics and an intraspecies UF of 10.

[1]US EPA (2016a,b) also evaluated human and animal studies on the carcinogenicity of PFOS and concluded that "there is *Suggestive Evidence of Carcinogenic Potential* of PFOS in humans" based on the liver and thyroid adenomas observed in the Butenhoff et al. (2012a) study. However, US EPA did not derive a cancer potency factor for PFOS. While the Butenhoff et al. (2012a) study reported statistically significant increased incidences of hepatocellular adenomas and carcinomas in male and female rats exposed to PFOS in the highest dose groups, as well as positive statistical trends for both datasets, US EPA (2016b) concluded that "existing evidence does not support a strong correlation between the tumor incidence and dose to justify a quantitative assessment."

[2]The dose associated with an excess lifetime cancer risk of one-in-one million (i.e., 1×10^{-6} dose), where, 1×10^{-6} dose = 1×10^{-6}/cancer potency factor.

[3]Health Canada (2018) calculated a chemical specific pharmacokinetic adjustment factor of 10 based on differences in PBPK modeled steady-state plasma PFOS predictions at 0.1 mg/kg-day between humans and rats [i.e., chemical specific UF = human steady state PFOS plasma level (360 micrograms per milliliter (mcg/mL)) ÷ estimated rat steady state PFOS plasma level (36.9 mcg/mL) = 10].

Table 4. Reference Doses for PFOS Derived by Authoritative Bodies.

Agency[1]	Reference Dose[2] (mg/kg-day)	Point of Departure		UF	Summary
		Dose (mg/kg-day) or Serum Concentration (mg/L)	Basis		
US EPA (2016a,b)	2.0×10^{-5}	6.26 mg/L in serum (rats) HED_{NOEL}= 0.00051 mg/kg-day	serum NOEL	30	Based on reduced body weight in offspring of rats exposed by gavage in a two-generation study. UF of 30: 10 for intraspecies differences and 3 for interspecies differences (Exhibit 1).
MDH (2019)	3.1×10^{-6}	2.36 mg/L in serum (mice) HED_{NOEL}= 0.000307 mg/kg-day	serum NOEL	100	Based on increased interleukin 4 (IL-4) and decreased sheep red blood cell (SRBC) specific IgM levels in adult male mice. UF of 100: 10 for intraspecies differences, 3 for interspecies differences, 3 for database uncertainties (Exhibit 2).
NJ DEP (2019)	1.8×10^{-6}	0.674 mg/L in serum (mice) HED_{NOEL} =0.000055 mg/kg-day	serum NOEL	30	Based on decreased plaque forming cell response in mice in a 60-day study. UF of 30: 10 for intraspecies differences and 3 for interspecies differences (Exhibit 3).

[1]The European Food Safety Authority Panel on Contaminants in the Food Chain (EFSA CONTAM) derived a tolerable weekly intake of 13 ng/kg-week for PFOS (equivalent to 1.8 ng/kg-day) based on increased total serum cholesterol in human epidemiological studies as part of a scientific opinion on the risks of PFOS in food. There is not a clear consensus among health agencies on whether cross-sectional studies such as those used by EFSA CONTAM in a weight of evidence approach provide sufficient evidence to establish causality, and whether the study limitations preclude their use for quantitative risk assessment (NJ DWQI, 2017; ATSDR 2018). Limitations in the approach used by EFSA included use of data packaged in quantiles rather than raw data points for benchmark dose modeling, and no adjustments for co-exposures to other perfluoroalkyl compounds. Based on these considerations, the EFSA derivation was not considered further as a basis for a potential ambient water quality value.

[2]Several agencies, including the Alaska Department of Environmental Conservation (2018), Connecticut State Department of Public Health (2016), Maine Center for Disease and Prevention (2017), Massachusetts Department of Environmental Protection (2018), Michigan Department of Environmental Quality (2018), and the Vermont Department of Health (2018) use the US EPA RfD and/or lifetime health advisory to define a health-based guidance value for PFOS in drinking water.

US EPA (2016a,b) REFERENCE DOSE DERIVATION AND LIFETIME DRINKING WATER HEALTH ADVISORY FOR PERFLUOROOCTANE SULFONATE (PFOS).

Source: US EPA (United States Environmental Protection Agency). 2016b. Drinking Water Health Advisory for Perfluorooctane Sulfonate (PFOS). Office of Water. EPA 822-R-16-004. Last accessed (03/25/2019) at https://www.epa.gov/ground-water-and-drinking-water/supporting-documents-drinking-water-health-advisories-pfoa-and-pfos.

5 DOSE-RESPONSE ASSESSME

As an initial step in the dose-respon

body weight changes in adults and offs
developmental effects (e.g., survival ar
selected based on their NOAEL and/or
two or more doses. From these studies,
(i.e., determination of HEDs) were sele
amenable for use in derivation of HED
pharmacokinetic model is limited beca
values for model input, as well as expo
to steady-state projections or applicabl
following short-term exposures. The pl
the animal studies are restricted to the
PFOS intake.

As described in section 3.2.4, EPA
to derive the average serum concentrat
from the toxicological database. Studie
demonstrated dose response and were
(AUC) at the time of sacrifice were use
values at the time of sacrifice with con

The NOAEL, LOAEL, and effect
average serum values and the percent
Table 5-1.

Table 5-1. Human Equivalent

Study	Dosing duration days	NOAE mg/kg/
Seacat et al. (2003): male rat ↑ALT, ↑BUN	98	0.34
Luebker et al. (2005b): ↓ rat pup body weight	84	0.1
Luebker et al. (2005a): ↓ rat pup body weight	63	None
Luebker et al. (2005a): rat ↓ maternal body	63	0.4

5.1 Uncertainty Factors

An uncertainty factor for intraspec
variability in the responses within the
life stage, health status) and extrinsic
exposure. No information was availab
supports a factor other than 10.

An uncertainty factor for interspec
uncertainty in extrapolating from labo
The three-fold factor is applied to acc
and humans. The HEDs were derived
pharmacokinetic differences between

An uncertainty factor for LOAEL
PODs, except the LOAEL of 0.4 mg/k
Luebker et al. (2005a) study. A value
same effect was 0.1 mg/kg/day in the
was not used in the one-generation stu
0.4 mg/kg/day, demonstrating that the

5.2 RfD Determination

Table 5-2 provides the calculati
NOAEL or LOAEL average serum
serum values measures collected at
applied to each POD; Table 5-2 illu
impacted by the doses used in the s
species/gender studied; therefore, th
individual study characteristics, hel
humans. It is important to note the
and study durations evaluated.

Table 5-2. Candidate RfDs Deriv

POD	HED POD mg/kg/day
(Seacat et al. 2003): male rat NOAEL for ↑ALT,	0.0013

from 0.00002 to 0.00005 mg/kg/day acr
calculated from HED average serum val
is derived from reduced pup body weigl
derivation of the RfD for PFOS is the H
that represents approximately 30% of st
3 UF_A) was applied to the HED NOAEI
supported by the 0.00002 mg/kg/day va
one-generation Luebker et al. (2005a) st
neurodevelopmental effects in the Butei

Low body weights in neonates are a
problems that often manifest later in life
pharmacokinetic modeling identified 0.:
Wistar rat pups exposed during gestatio:
insulin resistance, problems with glucos
as adults. A similar effect on glucose ho
study by Wan et al. (2014) with a dose (
fat content. For animals receiving a high
neurodevelopmental effects in Butenhol

indoor air in residential, commercial,
paint, furniture, and other consumer
PFOSA precursors that metabolically
industrial use of PFOS, as well as its

PFOS has also been detected in s
homes, offices, and vehicles. Inciden
route, particularly for small children
in soils and surface waters can affect
products, fish, and particulates in the

In summary, based on the physic
PFOS, there are many potentially sig
in its 2000 Methodology (USEPA 20
ingestion exist; however, informatior
from all of these different sources (B
an RSC of 20% (0.20) for PFOS.

6.2 Lifetime Health Advisory

The lifetime HA for PFOS is cal

A Drinking Water Equivalent Le
that 100% of PFOS exposure comes

$$DWEL = \frac{0.0}{(}$$

Where:

RfD = 0.00002 mg/kg/day; b
where dams were expose
through gestation and lac
DWI/bw = 0.054 L/kg/day; 9
and indirect community v
USEPA 2011b).

The lifetime HA is calculated aft

effects serving as the basis for the RfD
(e.g., reduced ossification and accelera

EXHIBIT 2. PERFLUOROOCTANE SULFONATE (PFOS)

MDH (2019) DERIVATION OF REFERENCE DOSE AND HEALTH-BASED WATER VALUE FOR PERFLUOROOCTANE SULFONATE

Source: MDH (Minnesota Department of Health). 2019. Toxicological Summary for: Perfluorooctane Sulfonate. Last accessed (04/04/2019) at https://www.health.state.mn.us/communities/environment/risk/docs/guidance/gw/pfos.pdf.

Toxicological Summary for

CAS: 45298-90-6 (anion)
1763-23-1 (acid)
29081-56-9 (ammonium salt)
70225-14-8 (diethanolamine salt)
2795-39-3 (potassium salt)
29457-72-5 (lithium salt)

Synonyms: PFOS, Perfluorooctane sulfo

MDH conducted a focused re-evaluatio

Reference Dose/Concentratio

Source of toxicity valu
Point of Departure (POD

Dose Adjustment Factor (DAF

Human Equivalent Dose (HED

Total uncertainty factor (UF
Uncertainty factor allocatio

Toxicokinetic Model Description (Goe

PFOS is well absorbed and is not meta
dose and clearance rate using the foll

Serum Concen

Where:
Dose (mg/kg-day) = Water or Breastn
and
Clearance (L/kg-d) = Volume of distrit

Two exposure scenarios were examin
contaminated water starting at birth a
life; and 2) an infant exclusively breas
water. In both scenarios the simulated
through placental transfer of PFOS (m

Summary of Reasonable Maximum Exp

Model Parameter	
Half-life	1241 d (5th to 9
Volume of distribution (Vd)	0.23 L/
Vd Age Adjustment Factor	2.1 age 10 yea
Clearance Rate (CR)	0.0001
Placental transfer factor *(% of maternal serum level)*	40% (n the lite (Mean
Breastmilk transfer factor *(% of maternal serum level)*	1.7% (report (No 95t
Water Intake Rate (L/kg-d)	95th pe 1 & 3-
Breastmilk Intake Rate (L-kg-d)	Upper

critical to note that background expo
while MDH's model predicts serum c
water source over time.

The apportionment to water ingestio
subtracting a conservative (high-end)
Eighty percent of the serum concentr
0.8). Subtracting the 95th percentile s
2018) as non-water background expo
leaves a residual serum concentratio
water. This residual concentration is
(24 µg/L) and approximately 54% of t
RSC of 50% for infants and young chil

Since exposures take years to elimina
steady-state serum levels in older ag
state conditions the 95th percentile (1
(Nelson 2018)) was used to determin

Figure 1. Formula-fed infant scenario
and an RSC of 50% for infants and you

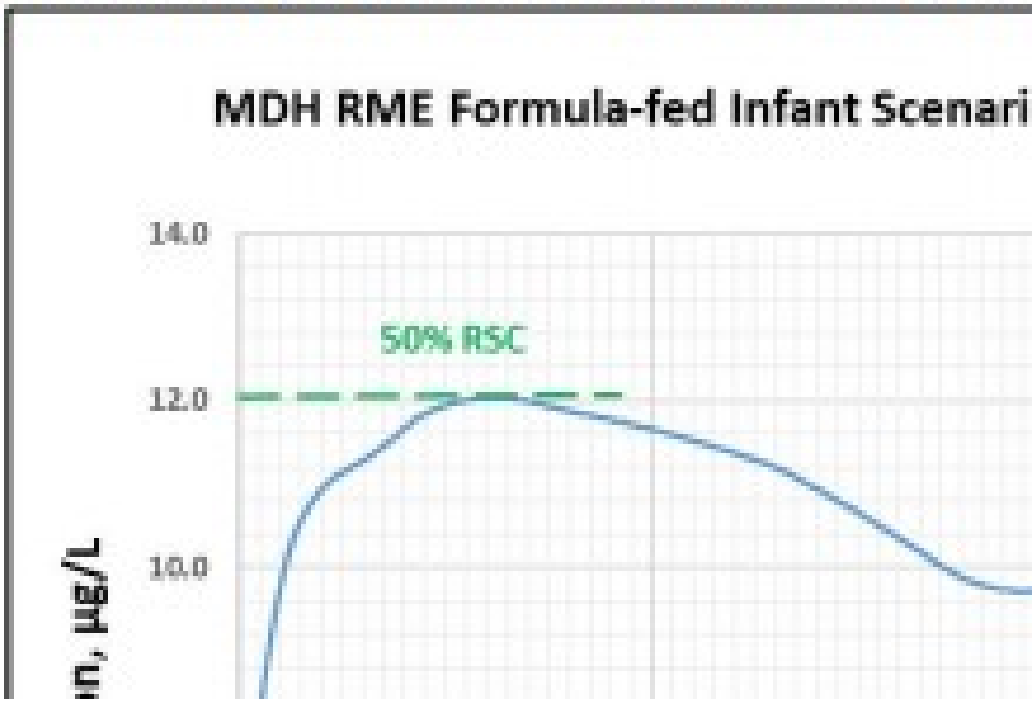


Figure 2. Formula-fed infant scenario se
and an RSC of 20% for steady-state.

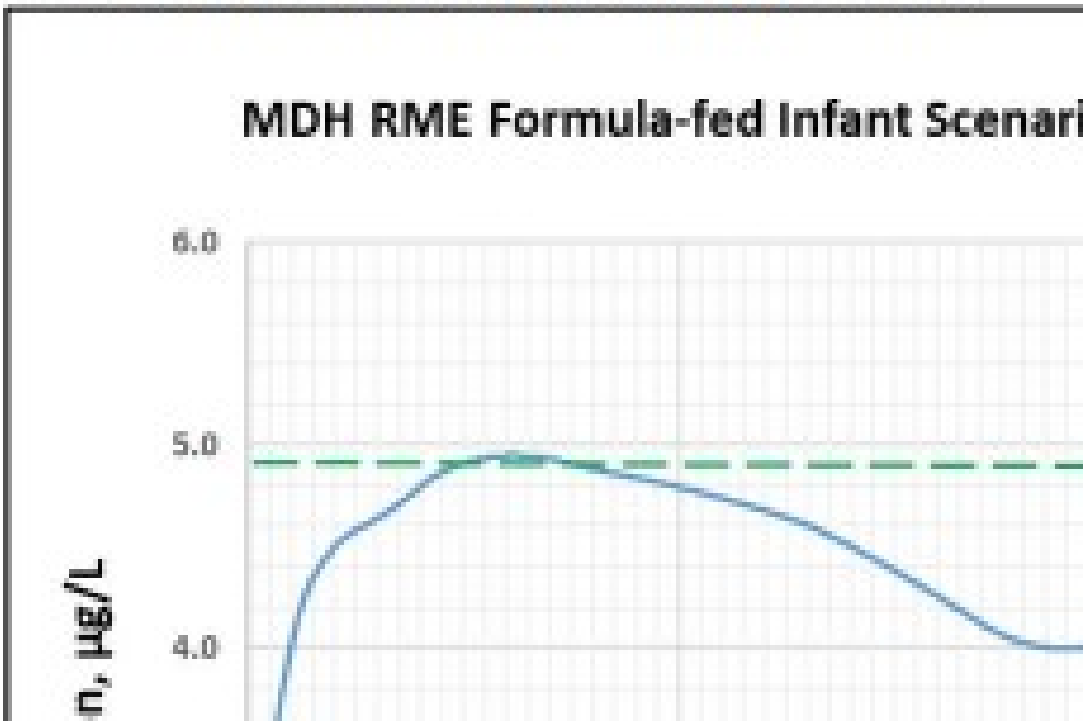
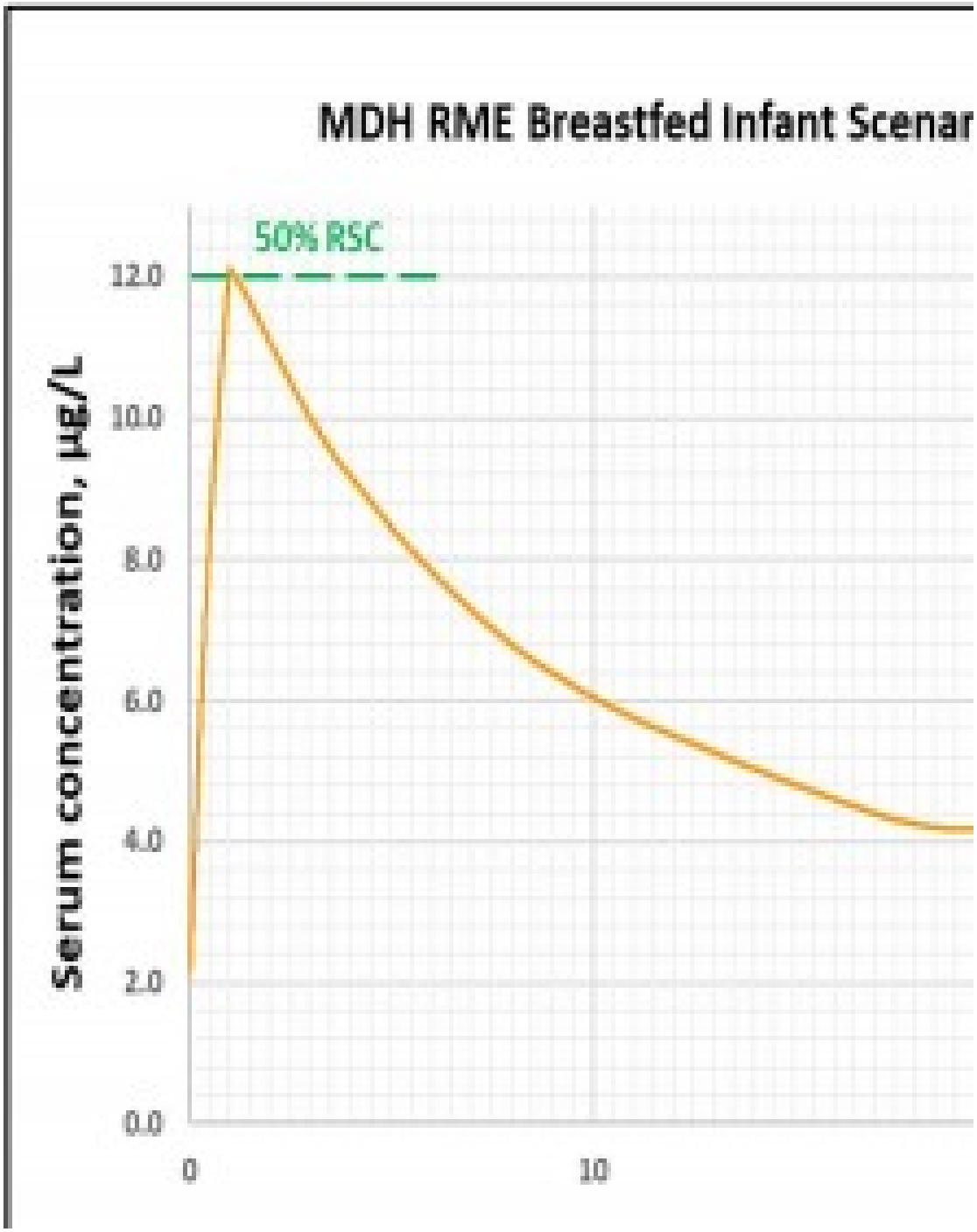


Figure 3. Breast-fed infant scenario se
and an RSC of 50% for infants and you



NJ DWQI (2018) HEALTH-BASED MCL FOR PERFLUOROOCTANE SULFONATE

Source: NJ DWQI (New Jersey Drinking Water Quality Institute). 2018. Health-Based Maximum Contaminant Level Support Document: Perfluorooctane Sulfonate (PFOS). Health Effects Subcommittee. Last accessed (03/21/2019) at https://www.state.nj.us/dep/watersupply/pdf/pfos-recommendation-appendix-a.pdf.

DEVELOPMENT OF POTENTIAL HI ENDPOINTS

The overall process used to develop potent endpoints is shown in Figure 15 and is dis PFOS are based on serum PFOS levels rat applied to the serum level PODs to develo Reference Doses (RfDs) but in terms of se Human Serum Levels are converted to Ref administered doses to human serum levels application of exposure factors for body w Relative Source Contribution factor to acc

Table 38. PODs, NOAELs and L endpoints identified for dose-resp	
Study	***Endpoint***
Butenhoff et al. (2012)	Hepatocellul hypertrophy rats)
Dong et al. (2009)	Relative live weight increa (male mice)
Dong et al. (2012a)	Relative live weight increa (male mice)
Dong et al. (2009)	Decreased pl forming imm response (male mice)

[a] Based on AUC

specific factors for which there is uncerta
of sensitive human sub-populations over
factors of 1 (no adjustment), 3 or 10, with
individual UFs represent log-units, the pr
UFs are considered in all cases:

$UF_{sub\text{-}chronic}$ – Applied to a sub-ch
NOAEL for a chronic duration stu
an exposure of > 30 day to ≤ 90 d

UF_{LOAEL} – Applied to an animal I
corresponding NOAEL, when no
The UF_{LOAEL} has the value of 1 in

Decreased plaque forming cell response (mal

$UF_{sub\text{-}chronic} = 1$

A sub-chronic to chronic unce
sub-chronic POD to account f
durations. The mice in Dong e
a subchronic duration (i.e., >
used because, as discussed in
cell response based on serum
to 60 days did not show a grea
below). In summary, this ind

Table 40. Calculation of Target Hu	
Study	***Animal i*** *(ng/ml*
Butenhoff et al. (2012) (Hepatocellular hypertrophy)	4,5
Dong et al. (2012a) (Increased relative liver weight)	4,3
Dong et al. (2009) (Decreased plaque forming cell response)	6

Calculation of RfDs from Target Human S

The RfD (as an intake dose; mg/kg/day) is ca

Exposure factors for Health-based M

The Health-based MCL is a PFOS drink
drinking water consumption over a lifeti
RfD for decreased plaque forming cell r
kg), daily drinking water ingestion (2 L/
(20%; discussed below).

Relative Source Contribution (RSC) I

A Relative Source Contribution (RSC) f
including food, soil, air, water, and cons
and other states in the development of h
carcinogenic effects. The RSC is intend
exceeding the RfD (USEPA, 2000b). W
drinking water exposures is not availabl
that 20% of exposure comes from drinki
chemical-specific exposure data are avai
derived, with floor and ceiling RSC valu

than older individuals. Infants consume m
individuals on a body weight basis and, P
similar or higher than in the mother's drin

These higher infant exposures must be co
sensitive toxicological effect occurred fro
exposures in infancy. The dose-response
plaque forming cells in mice (an indicator
vaccine response in humans) was similar
durations, indicating that the Reference D
as well as chronic exposures.

For the reasons discussed above, the defa
based MCL.

Derivation of potential Health-based M

The equation used to derive the Health-ba

List of Abbreviations and Acronyms Frequently Used in New York State Human Health Fact Sheets.

1×10^{-6}	one-in-one million
ACPF	adjusted cancer potency factor
ADAF	age-dependent adjustment factor
ADI	acceptable daily intake
adj	adjusted
AIC	Akaike information criterion
ATSDR	Agency for Toxic Substance and Disease Registry
AUC	area under the curve
AWQGV	ambient water quality guidance value
BMC	benchmark concentration
BMCL	benchmark concentration, lower 95% confidence limit
BMD	benchmark dose
BMDL	benchmark dose, lower 95% confidence limit
$BMDL_{10}$	BMDL, 10% BMR
$BMDL_{50}$	BMDL, 50% BMR
$BMDL_{1SD}$	BMDL, BMR of one standard deviation
$BMDL_{ADJ}$	BMDL, adjusted to continuous exposure
BMR	benchmark response
BW	body weight
$BW^{2/3}$	body-weight raised to the 2/3 power scaling
$BW^{3/4}$	body-weight raised to the 3/4 power scaling
CA EPA	California Environmental Protection Agency
CASRN	Chemical Abstracts Service Registry Number
CDC	Centers for Disease Control and Prevention
CI	confidence interval
CL	confidence limit
CNS	central nervous system
CPF	cancer potency factor
DAF	dosimetric adjustment factor
DNA	deoxyribonucleic acid
DWCR	drinking water consumption rate
EFSA	European Food Safety Authority
F_1	first filial generation (in experimental animals)
F_2	second filial generation (in experimental animals)
FAO	Food and Agriculture Organization of the United Nations
FIFRA	Federal Insecticide, Fungicide, and Rodenticide Act
g	gram
GD	gestation day
HC	Health Canada
HEC	human equivalent concentration
HED	human equivalent dose
HED_{BMDL10}	human equivalent dose at the $BMDL_{10}$
HED_{LOEL}	human equivalent dose at the LOEL
HED_{NOEL}	human equivalent dose at the NOEL
HI	hazard index

hr	hour
HSDB	Hazardous Substance Data Bank
IARC	International Agency for Research on Cancer
IRIS	Integrated Risk Information System, US EPA
kg	kilogram
L	liter
L/day	liters per day
L/kg	liters per kilogram
L/kg-day	liters per kilogram day
LADC	lifetime average daily concentration
LADD	lifetime average daily dose
LCL	lower confidence limit
LED	lower bound on effective dose
LEL	lowest-effect level
LOAEL	lowest-observed-adverse-effect level
LOEL	lowest-observed-effect level
mcg	microgram
mcg/m^3	micrograms per cubic meter
mcg/kg-day	micrograms per kilogram body weight per day
mcg/L	micrograms per liter
MCL	maximum contaminant level
MCLG	maximum contaminant level goal
MDPH	Massachusetts Department of Public Health
mg	milligram
mg/kg	milligrams per kilogram
mg/L	milligrams per liter
mg/hr	milligrams per hour
mg-hr/L	milligrams-hour per liter
mg/kg-day	milligrams per kilogram body weight per day
mg/kg/day	milligrams per kilogram body weight per day
mg/m^3	milligrams per cubic meter
MLE	maximum likelihood estimate
MOA	mode-of-action
MRL	minimal risk level
MTD	maximum tolerated dose
NAS	National Academy of Sciences
NHANES	National Health and Nutrition Examination Survey
ng	nanogram
ng/L	nanograms per liter
NOAEL	no-observed-adverse-effect level
NOEL	no-observed-effect level
NRC	National Research Council
NTP	National Toxicology Program
NYS	New York State
NYS DEC	New York State Department of Environmental Conservation
NYS DOH	New York State Department of Health
NYCRR	New York Code of Rules and Regulations
OPP	Office of Pesticide Programs, US EPA
P (value)	probability value
PBPK	physiologically-based pharmacokinetic

PDAF	pharmacodynamic adjustment factor
pg	picogram
pg/L	picograms per liter
PKAF	pharmacokinetic adjustment factor
POC	principal organic contaminant
POD	point-of-departure
ppb	parts per billion
ppm	parts per million
ppt	parts per trillion
RfC	reference concentration
RfD	reference dose
RPF	relative potency factor
RR	relative risk
RSC	relative source contribution
SAB	EPA Science Advisory Board
SD	standard deviation
TDI	tolerable daily intake
TEF	toxic equivalency factor
TEQ	toxicity equivalent
TW	time-weighted
TWA	time-weighted-average
UCL	upper confidence limit
UCMR	Unregulated Contaminant Monitoring Rule, US EPA
UF	uncertainty factor
UOC	unspecified organic contaminant
UR	unit risk
U.S.	United States
US EPA	United States Environmental Protection Agency
WBC	white blood cell
WCAF	water consumption adjustment factor
WHO	World Health Organization
wk	week