EPA/635/R-23/027Fa
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IRIS Toxicological Review of Perfluorohexanoic Acid [PFHxA, CASRN 307-24-4] and Related Salts

April 2023

Integrated Risk Information System
Center for Public Health and Environmental Assessment
Office of Research and Development
U.S. Environmental Protection Agency
Washington, DC

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ABBREVIATIONS AND ACRONYMS

ADME	absorption, distribution, metabolism, and excretion
AFFF	aqueous film-forming foam
A:G	albumin:globulin ratio
AIC	Akaike's information criterion
ALP	alkaline phosphatase
ALT	alanine aminotransferase
APTT	activated partial thromboplastin time
AST	aspartate aminotransferase
atm	atmosphere
ATSDR	Agency for Toxic Substances and Disease Registry
AUC	area under the curve
BMD	benchmark dose
BMDL	benchmark dose lower confidence limit
BMDS	Benchmark Dose Software
BMR	benchmark response
BUN	blood urea nitrogen
BW	body weight
C_{max}	maximum concentration
CAR	constitutive androstane receptor
CASRN	Chemical Abstracts Service registry number
CBC	complete blood count
CI	confidence interval
CL	clearance
CL_A	clearance in animals
CL_H	clearance in humans
CPHEA	Center for Public Health and Environmental Assessment
CPN	chronic progressive nephropathy
DAF	dosimetric adjustment factor
DNA	deoxyribonucleic acid
DTXSID	DSSTox substance identifier
eGFR	estimated glomerular filtration rate
EPA	Environmental Protection Agency
ER	extra risk
FTOH	fluorotelomer alcohol
GD	gestation day
GGT	γ-glutamyl transferase
HAWC	Health Assessment Workplace Collaborative
HCT	hematocrit
HED	human equivalent dose
HERO	Health and Environmental Research Online
HGB	hemoglobin
HSA	human serum albumin
IQR	interquartile range
IRIS	Integrated Risk Information System
ISI	Influential Scientific Information
IUR	inhalation unit risk
i.v.	intravenous
LDH	lactate dehydrogenase
LOQ	limit of quantitation
LOAEL	lowest-observed-adverse-effect level
LOD	limit of detection
LOEC	lowest-observed-effect concentration
MCH	mean cell hemoglobin
MCHC	mean cell hemoglobin concentration
MCV	mean cell volume
MOA	mode of action
MW	molecular weight
NCTR	National Center for Toxicological Research
NOAEL	no-observed-adverse-effect level
NPL	National Priorities List
NTP	National Toxicology Program
ORD	Office of Research and Development
OECD	Organisation for Economic Co-operation and Development
OSF	oral slope factor
osRfD	organ/system-specific oral reference dose
PBPK	physiologically based pharmacokinetic
PC	partition coefficient
PECO	populations, exposures, comparators, and outcomes
PFAA	perfluoroalkyl acids
PFAS	per- and polyfluoroalkyl substances
PFBA	perfluorobutanoic acid
PFBS	perfluorobutane sulfonate
PFCA	perfluorinated carboxylic acid
PFDA	perfluorodecanoic acid
PFHxA	perfluorohexanoic acid
PFHxS	perfluorohexane sulfonate
PFNA	perfluorononanoic acid
PFOA	perfluorooctanoic acid
PFOS	perfluorooctane sulfonate
PK	pharmacokinetic
PND	postnatal day
POD	point of departure
POD_{HED}	human equivalent dose POD
PPAR	peroxisome proliferated activated receptor
PQAPP	programmatic quality assurance project plan
PT	prothrombin time
QA	quality assurance
QAPP	quality assurance project plan
QMP	quality management plan
RBC	red blood cells

RD	relative deviation
RfC	reference concentration
RfD	oral reference dose
RNA	ribonucleic acid
ROS	reactive oxygen species
RXR	retinoid X receptor
SD	standard deviation
TP	total protein
TRI	Toxics Release Inventory
TSCATS	Toxic Substances Control Act Test Submissions
TSH	thyroid stimulating hormone
UF	uncertainty factor
UF_A	interspecies uncertainty factor
UF_C	composite uncertainty factor
UF_D	evidence base deficiencies uncertainty factor
UF_H	human variation uncertainty factor
UF_L	LOAEL to NOAEL uncertainty factor
UF_S	subchronic to chronic uncertainty factor
V_2	volume of distribution of peripheral compartment (two-compartment PK model)
V_d	volume of distribution

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EXECUTIVE SUMMARY

Summary of Occurrence and Health Effects

Perfluorohexanoic acid (PFHxA, CASRN 307-24-4)[1] and its related salts are members of the group per and polyfluoroalkyl substances (PFAS). This assessment applies to PFHxA as well as salts of PFHxA, including ammonium perfluorohexanoate (PFHxA-NH_4, CASRN 21615-47-4), and sodium perfluorohexanoate (PFHxA-NA, CASRN 2923-26-4), and other nonmetal and alkali metal salts of PFHxA that would be expected to fully dissociate in aqueous solutions of pH ranging from 4–9 (e.g., in the human body) and not release other moieties that would cause toxicity independent of PFHxA. Notably, due to the possibility of PFHxA-independent contributions of toxicity, this assessment would not necessarily apply to nonalkali metal salts of PFHxA (e.g., silver perfluorohexanoate; CASRN 336-02-7). The synthesis of evidence and toxicity value derivation presented in this assessment focuses on the free acid of PFHxA and related ammonium and sodium salts given the currently available toxicity data.

Concerns about PFHxA and other PFAS stem from the resistance of these compounds to hydrolysis, photolysis, and biodegradation, which leads to their persistence in the environment. PFAS are not naturally occurring in the environment; they are manmade compounds that have been used widely over the past several decades in industrial applications and consumer products because of their resistance to heat, oil, stains, grease, and water. PFAS in the environment are linked to industrial sites, military fire training areas, wastewater treatment plants, and commercial products (Appendix A, Section 2.1.2)

The Integrated Risk Information System (IRIS) Program is developing a series of five PFAS assessments (i.e., perfluorobutanoic acid [PFBA], perfluorohexanoic acid [PFHxA], perfluorohexane sulfonate [PFHxS], perfluorononanoic acid [PFNA], perfluorodecanoic acid [PFDA], and their associated salts) at the request of EPA National Programs and Regions. Specifically, the development of human health toxicity assessments for exposure to these individual PFAS represents only one component of the broader PFAS strategic roadmap at the EPA (https://www.epa.gov/pfas/pfas-strategic-roadmap-epas-commitments-action-2021-2024). The systematic review protocol (see Appendix A) for these five PFAS assessments outlines the related scoping and problem formulation efforts, including a summary of other federal and state assessments of PFHxA. The protocol also lays out the systematic review and dose-response methods used to conduct this review (see also Section 1.2). The systematic review protocol was released for public comment in November 2019 and was updated based on those public comments. Appendix A links to the updated version of the protocol and summary of revisions.

[1] The CASRN given is for linear PFHxA; the source PFHxA used in toxicity studies was reported to be >93% pure. No explicit statement that only the linear form was used was available from the studies. Therefore, there is the possibility that a minor proportion of the PFHxA used in the studies were branched isomers and thus observed health effects may apply to the total linear and branched isomers in a given exposure source.

Human epidemiological studies have examined possible associations between PFHxA exposure and health outcomes, such as liver enzymes, thyroid hormones, blood lipids, blood pressure, insulin resistance, body mass index, semen parameters, reproductive hormones, and asthma. The ability to draw conclusions regarding these associations is limited by the overall conduct of the studies (studies were generally *low* confidence); the few studies per health outcome; and, in some studies, the lack of a quantifiable measure of exposure. No studies were identified that evaluated the association between PFHxA exposure and carcinogenicity in humans.

Animal studies of PFHxA exposure exclusively examined the oral exposure route, and therefore, no inhalation assessment was conducted nor was an RfC derived (see Section 5.2.2). The available animal studies of oral PFHxA exposure examined a variety of noncancer and cancer endpoints, including those relevant to hepatic, developmental, renal, hematopoietic, endocrine, reproductive, immune, and nervous system effects.

Overall, the available **evidence indicates** that PFHxA likely causes hepatic, developmental, hematopoietic, and endocrine (see Sections 3.2.1, 3.2.2, 3.2.4, and 3.2.5, respectively) effects in humans given sufficient exposure conditions. Specifically, for hepatic effects, the primary support for this hazard conclusion included evidence of increased relative liver weights and increased incidence of hepatocellular hypertrophy in adult rats. These hepatic findings correlated with changes in clinical chemistry (e.g., serum enzymes, blood proteins) and necrosis. For hematopoietic effects, the primary supporting evidence included decreased red blood cell counts, decreased hematocrit values, and increased reticulocyte counts in adult rats. Developmental effects were identified as a hazard based on evidence of decreased offspring body weight and increased perinatal mortality in exposed rats and mice. A short-term (28-day) study in rats showed a strong dose dependent effect on serum thyroid hormones in males. Selected quantitative data from these identified hazards were used to derive toxicity values (see Table ES-1).

Although some human and animal evidence was also identified for renal, male, and female reproductive, immune, and nervous system effects, the currently available ***evidence is inadequate*** to assess whether PFHxA may cause these health effects in humans (see Sections 3.2.3, 3.2.6, 3.2.7, 3.2.8, and 3.2.9 respectively) and were not used to derive toxicity values.

Table ES-1. Evidence integration judgments and derived toxicity values for PFHxA

Health system	Evidence integration judgment	Toxicity value type	Value for PFHxA (mg/kg-d)	Value for PFHxA-Na[a] (mg/kg-d)	Value for PFHxA-NH_4[a] mg/kg-d)	Confidence in toxicity value[b]	UF_C[c,d,e]	Basis
Hepatic	*Evidence indicates (likely)*	osRfD	4×10^{-4}	4×10^{-4}	4×10^{-4}	*Medium*	300	Increased hepatocellular hypertrophy in adult rats (Loveless et al., 2009)
		Subchronic osRfD	1×10^{-3}	1×10^{-3}	1×10^{-3}	*Medium*	100	Increased hepatocellular hypertrophy in adult rats (Loveless et al., 2009)
Hematopoietic	***Evidence indicates (likely)***	osRfD	5×10^{-3}	6×10^{-3}	5×10^{-3}	*Medium*	100	Decreased red blood cells in adult rats (Klaunig et al., 2015)
		Subchronic osRfD	8×10^{-4}	8×10^{-4}	8×10^{-4}	*Medium-Low*	100	Decreased red blood cells in adult rats (Chengelis et al., 2009b)
Developmental	***Evidence indicates (likely)***	osRfD	5×10^{-4}	5×10^{-4}	5×10^{-4}	*Medium*	100	Decreased F_1 body weight at PND 0 in rats (Loveless et al., 2009)
		Subchronic osRfD	5×10^{-4}	5×10^{-4}	5×10^{-4}	*Medium*	100	Decreased F_1 body weight at PND 0 in rats (Loveless et al., 2009)
Endocrine	***Evidence indicates (likely)***	osRfD	NA	NA	NA	NA	NA	Not derived due to high degree of uncertainty with deriving a lifetime value from a short-term study.
		Subchronic osRfD	1×10^{-3}	1×10^{-3}	1×10^{-3}	*Medium*	300	Decreased Free T4 in adult male rats (NTP, 2018)
RfD[d]			5×10^{-4}	5×10^{-4}	5×10^{-4}	*Medium*	100	Decreased F_1 body weight at PND 0 in rats (Loveless et al., 2009)
Subchronic RfD[e]			5×10^{-4}	5×10^{-4}	5×10^{-4}	*Medium*	100	Decreased F_1 body weight at PND 0 in rats (Loveless et al., 2009)

See Section 5.2.1 for full details on study and dataset selection, modeling approaches (including BMR selection), uncertainty factor application, candidate value selection, and characterization of confidence in the osRfDs and RfDs.

RfD = reference dose (in mg/kg-day) for lifetime exposure; subchronic RfD = reference dose (in mg/kg-d) for less-than-lifetime exposure; osRfD = organ/system specific oral reference dose (in mg/kg-d); UF_C = composite uncertainty factor which is the product of the interspecies uncertainty factor (UF_A), interindividual human variability uncertainty factor (UF_H), subchronic-to-chronic uncertainty factor (UF_S), LOAEL-to-NOAEL uncertainty factor (UF_L), and database uncertainty factor (UF_D); NA = not applicable.

[a]See Tables 5-7 and 5-11 for details on how to calculate candidate values for salts of PFHxA. The osRfDs presented in this table have been rounded to 1 significant digit from the candidate values presented in Tables 5-7 and 5-11.

[b]The overall confidence in the derived toxicity values is synthesized from confidence judgments regarding confidence in the study used to derive the toxicity value, confidence in the evidence base supporting the hazard, and confidence in the quantification of the point of departure; see Table 5-8 for full details regarding the confidence judgments.

[c]See Table 5-6 for an explanation of the uncertainty factors applied to derive the osRfD and subchronic osRfD values.

[d]Developmental and hematopoietic $UF_C = 100$ based on $UF_A = 3$, $UF_H = 10$, $UF_S = 1$, $UF_L = 1$, and $UF_D = 3$; hepatic $UF_C = 300$ based on $UF_A = 3$, $UF_H = 10$, $UF_S = 3$, $UF_L = 1$, and $UF_D = 3$.

[e]Hepatic, developmental, and hematopoietic $UF_C = 100$ based on $UF_A = 3$, $UF_H = 10$, $UF_S = 1$, $UF_L = 1$, and $UF_D = 3$; endocrine $UF_C = 300$ based on $UF_A = 3$, $UF_H = 10$, $UF_S = 3$, $UF_L = 1$, and $UF_D = 3$.

ES.1 CHRONIC ORAL REFERENCE DOSE (RFD) FOR NONCANCER EFFECTS

From the identified hazards of potential concern (i.e., endocrine, hepatic, hematopoietic, and developmental toxicity), decreased offspring body weight in neonatal rats (Loveless et al., 2009) was selected as the basis for the RfD of 5×10^{-4} mg/kg-day. A $BMDL_{5RD}$ of 10.62 mg/kg-day was identified for this endpoint and was used as the point of departure (POD). The human equivalent dose POD (POD_{HED}) of 0.048 mg/kg-day was derived by applying the ratio of the clearance between female rats and humans and a normalization from the sodium salt to the free acid using a molecular weight conversion. The overall RfD for PFHxA was calculated by dividing the POD_{HED} by a composite uncertainty factor of 100 to account for pharmacodynamic uncertainty in the extrapolation from rats to humans (UF_A = 3), interindividual differences in human susceptibility (UF_H = 10), and deficiencies in the toxicity evidence base (UF_D = 3).

ES.2 CONFIDENCE IN THE ORAL REFERENCE DOSE (RFD)

The study conducted by Loveless et al. (2009) reported developmental effects following administration of PFHxA sodium salt to pregnant Sprague-Dawley rats dosed by gavage for approximately 70 days prior to cohabitation through gestation and lactation, for a total of 126 days daily gavage with 0, 20, 100, or 500 mg/kg-day sodium PFHxA. The overall confidence in the osRfD is medium and is primarily driven by medium confidence in the overall evidence base for developmental effects, high confidence in the study (click the HAWC link for full study evaluation details), and medium confidence in quantitation of the POD (see Table 5-8). High confidence in the study was not interpreted to warrant changing the overall confidence in the RfD from medium.

ES.3 SUBCHRONIC ORAL REFERENCE DOSE (RFD) FOR NONCANCER EFFECTS

In addition to providing RfDs for chronic oral exposures in multiple systems, a less-than-lifetime subchronic RfD was derived for PFHxA. The same study and endpoint (Loveless et al., 2009) and decreased F_1 body weight and value was selected as the basis for the subchronic RfD of 5×10^{-4} mg/kg-day (see Table ES-1). Details are provided in Section 5.2.1.

ES.4 NONCANCER EFFECTS FOLLOWING INHALATION EXPOSURE

No studies that examine toxicity in humans or experimental animals following inhalation exposure and no physiologically based pharmacokinetic (PBPK) models are available to support route-to-route extrapolation; therefore, no RfC was derived.

ES.5 EVIDENCE FOR CARCINOGENICITY

Under EPA's *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005), EPA concluded there is *inadequate information to assess carcinogenic potential* for PFHxA by all routes of exposure. The lack of data on the carcinogenicity of PFHxA precludes the derivation of quantitative estimates for either oral (oral slope factor [OSF]) or inhalation (inhalation unit risk [IUR]) exposure (see Section 3.3).

1. OVERVIEW OF BACKGROUND INFORMATION AND ASSESSMENT METHODS

A series of five PFAS assessments (perfluorobutanoic acid [PFBA], perfluorohexanoic acid [PFHxA], perfluorohexane sulfonate [PFHxS], perfluorononanoic acid [PFNA], perfluorodecanoic acid [PFDA], and their associated salts) are being developed by the Integrated Risk Information System (IRIS) Program at the request of the U.S. Environmental Protection Agency (EPA) National Programs and Regions. Appendix A is the systematic review protocol for these five PFAS assessments. The protocol outlines the scoping and problem formulation efforts relating to these assessments, including a summary of other federal and state reference values for PFHxA. The protocol also lays out the systematic review and dose-response methods used to conduct this review (see also Section 1.2). This systematic review protocol was released for public comment in November 2019 and was subsequently updated based on those public comments. Appendix A includes the updated version of the protocol, including a summary of the updates in the protocol history section (see Appendix A, Section 12).

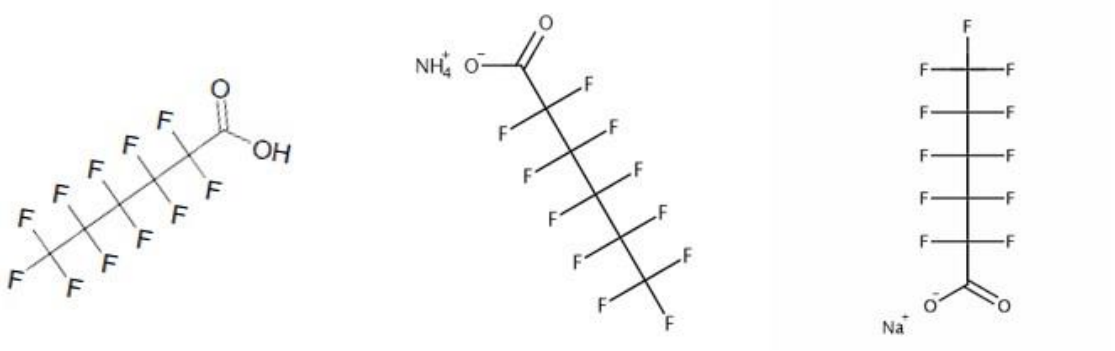
1.1. BACKGROUND INFORMATION ON PFHxA AND RELATED AMMONIUM AND SODIUM SALTS

This section provides a brief overview of aspects of the physiochemical properties, human exposure, and environmental fate characteristics of perfluorohexanoic acid (PFHxA, CASRN 307-24-4), ammonium perfluorohexanoate (PFHxA-NH_4, CASRN 21615-47-4), and sodium perfluorohexanoate (PFHxA-Na, CASRN 2923-26-4). This overview is not intended to provide a comprehensive description of the available information on these topics and is not recommended for use in decision making. The reader is encouraged to refer to source materials cited below, more recent publications on these topics, and the assessment systematic review protocol (see Appendix A).

1.1.1. Physical and Chemical Properties

PFHxA and related sodium and ammonium PFHxA salts covered in this assessment are members of the group of per- and polyfluoroalkyl substances (PFAS). Concerns about PFHxA and other PFAS stem from the resistance of these compounds to hydrolysis, photolysis, and biodegradation, which leads to their persistence in the environment (NLM, 2017, 2016, 2013). PFHxA and related salts are classified as a perfluorinated carboxylic acids (PFCAs) (OECD, 2015). PFHxA and its associated salts are considered short-chain PFAS (ATSDR, 2021). The linear chemical

structures[2] of these chemicals are presented in Figure 1-1, and select physiochemical properties are provided in Table 1-1. When available, experimental values are provided in the table but predicted values that may be less reliable are included in the absence of experimental data.



	PFHxA	PFHxA ammonium salt	PFHxA sodium salt
CASRN	307-24-4	21615-47-4	2923-26-4
DTXSID	3031862	90880232	3052856

Figure 1-1. Linear chemical structures of (from left to right) PFHxA, PFHxA NH_4, and PFHxA-Na.

PFHxA = perfluorohexanoic acid; PFHxA NH_4 = ammonium perfluorohexanoate; PFHxA-Na = sodium perfluorohexanoate.

Source: EPA CompTox Chemicals Dashboard.

Table 1-1. Physicochemical properties of PFHxA

Property (unit)	PFHxA value	PFHxA-NH_4 value	PFHxA-Na value
Formula	$C_6HF_{11}O_2$	$C_6H_4F_{11}NO_2$	$C_6F_{11}NaO_2$
Molecular weight (g/mol)	314	331	336
Melting point (°C)	12.2[a]	39.2[b]	70.2[b]
Boiling point (°C)	157[a]	156[b]	216[b]
Density (g/cm^3)	1.69[b]	1.72[b]	1.69[b]
Vapor pressure (mm Hg)	0.908[a]	2.00[b]	1.63[b]
Henry's law constant (atm-m^3/mole)	2.35×10^{-10} (b)	2.35×10^{-10} (b)	2.35×10^{-10} (b)
Water solubility (mol/L)	9.34×10^{-5} (a)	1.10[b]	8.78×10^{-5} (a)

[2]The assessment applies to other non-linear isomers of PFHxA and related salts.

Property (unit)	PFHxA value	PFHxA-NH_4 value	PFHxA-Na value
PKa	−0.16[c]	–	–
$LogP_{Octanol\text{-}Water}$[d]	2.85[a]	3.97[b]	0.70[a]
Soil adsorption coefficient (L/kg)	1,070[b]	1,070[b]	1,070[b]
Bioconcentration factor	49.3[b]	5.47[b]	49.3[b]

"–" = data not available.
[a](U.S. EPA, 2018a). CompTox Chemicals Dashboard; access date 2/18/2021. Median or average experimental values.
[b]Average or median predicted values are, in general, less reliable than experimental values.
[c]Predicted value reported by Steinle-Darling and Reinhard (2008).
[d]Experimental measure of PFAS octanol/water partition coefficient are difficult due to the tendency for alkyl acids to aggregate at the interface between octanol and water (Kim et al., 2015).

1.1.2. Sources, Production, and Use

PFAS have been used widely over the past several decades in consumer products and industrial applications because of their resistance to heat, oil, stains, grease, and water. (ATSDR, 2021; U.S. EPA, 2020, 2019c, 2013, 2007, 2002b). Fluorinated compounds have also been used in consumer products including stain-resistant fabrics for clothing, carpets, and furniture; nonstick cookware; ski wax; certain leather products; and personal care products (e.g., dental floss, cosmetics, and sunscreen) (ATSDR, 2021; U.S. EPA, 2020, 2019c, 2013, 2007, 2002b). PFAS also have been detected from foam used in firefighting and in industrial surfactants, emulsifiers, wetting agents, additives, and coatings; they are also used in aerospace, automotive, building, and construction industries to reduce friction (U.S. EPA, 2020, 2019c; ATSDR, 2018; U.S. EPA, 2013, 2007, 2002b). PFHxA has been detected as a breakdown product of PFAS used in water- and stain-protective coatings for carpets, paper, and textiles including textiles used in some protective clothing (Klaunig et al., 2015). PFAS have been found at private and federal facilities associated with various material or processes involving aqueous film-forming foam (AFFF), chrome plating, and are associated with other industries using PFAS (e.g., textiles, carpets) (ATSDR, 2021; U.S. EPA, 2020, 2019c, 2013, 2007, 2002b). In AFFF, PFHxA has been detected at concentrations ranging from 0.1 to 0.3 g/L (Baduel et al., 2015; Houtz et al., 2013). In the occupational sectors and/or products PFAS occurances have been found in wood particle board, rubber insulation, electroplating, metal treatments, paints, varnishes, and flame retardants (OECD, 2022; Glüge et al., 2020).

No quantitative PFHxA information on production volume is available (U.S. EPA, 2019a), and EPA's Toxics Release Inventory (TRI) contains no information on releases to the environment from facilities manufacturing, processing, or otherwise using PFHxA (ATSDR, 2021; U.S. EPA, 2018d).

Wang et al. (2014) estimates global emissions of 39 to 1,691 tons of PFHxA from direct and indirect (i.e., degradation of precursors) sources between 1951 and 2030. The lower estimate assumes manufacturers cease production and use of long-chain PFCAs and that their precursors

stay consistent with global transition trends. The higher estimate assumes the 2015 emission scenario remains constant until 2030.

1.1.3. Environmental Fate and Transport

PFAS are highly stable and persistent worldwide, and many are found in environmental media (e.g., soils, water, the atmosphere, foods, wildlife, and humans) (U.S. EPA, 2019c) (Appendix A).

Uptake of soil PFAS to plants can occur (ATSDR, 2021), and estimates are available of PFAS accumulation in vegetation when plants are grown in PFAS-contaminated soil. Yoo et al. (2011) estimated grass-soil accumulation factors of 3.4 (grass concentration divided by soil concentration) for PFHxA using samples collected from a site with biosolids-amended soil. Venkatesan and Halden (2014) analyzed archived samples from outdoor mesocosms to investigate the fate over 3 years of PFAS in agricultural soils amended with biosolids. The mean half-life for PFHxA was estimated to be 417 days. Volatilization of PFHxA from moist soil is not expected to be an important fate process (NLM, 2016). PFHxA bioaccumulates in foods grown on PFAS-containing soils. Blaine et al. (2013) conducted a series of greenhouse and field experiments to investigate the potential for PFAS uptake by lettuce, tomatoes, and corn when grown in industrially impacted and biosolids-amended soils. Blaine et al. (2013) calculated PFHxA bioaccumulation factors of 9.9–11.7 for lettuce and 2.9–6.8 for tomatoes (no bioaccumulation factor was reported for corn).

1.1.4. Potential for Human Exposure and Populations with Potentially Greater Exposure

The general population can be exposed to PFAS via inhalation of air or dust, ingestion of drinking water and food, and dermal contact with PFAS-containing products and during susceptible lifestages (see Appendix A). The oral route of exposure is considered the dominant exposure pathway for the general population (Sunderland et al., 2019), for which contaminated drinking water is likely a significant source of exposure to PFAS, including PFHxA. Due to the high water solubility and mobility of PFAS in groundwater (and potential lack of remediation at some water treatment facilities), populations consuming drinking water from any contaminated watershed could be exposed to PFAS (Shao et al., 2016).

Infants potentially have higher exposure due to greater ingestion of food per body weight. PFHxA has been detected in human breast milk from many nations, including U.S. (Zheng et al., 2021), French, Korean, and Spanish populations (summarized in Table 5 of Anderson et al. (2019)). Exposure can also occur through hand-to-mouth transfer of materials containing these compounds (ATSDR, 2021) or in infants through ingestion of formula reconstituted with contaminated drinking water.

Air and Dust

PFHxA has not been evaluated under the National Air Toxics Assessment program and no additional information on atmospheric concentration was identified. PFAS, including PFHxA, have

been measured in indoor air and dust and might be associated with the indoor use of consumer products such as PFAS-treated carpets or other textiles (ATSDR, 2021). For example, Kato et al. (2009) detected PFHxA in 46.2% of the dust samples collected from 39 homes in the United States, United Kingdom, Germany, and Australia. Karásková et al. (2016) detected PFHxA in all 56 dust samples collected from 41 homes in the Czech Republic, Canada, and the United States at mean concentrations of 12.8, 14.5, and 20.9 ng/g, respectively. Strynar and Lindstrom (2008) analyzed dust samples from 110 homes and 10 daycare centers in North Carolina and Ohio and detected PFHxA in 92.9% of the samples. Knobeloch et al. (2012) detected PFHxA in 20% of samples of vacuum cleaner dust collected from 39 homes in Wisconsin. PFHxA concentrations ranged from below the reporting limit (1 ng/g) to 180 ng/g. Fraser et al. (2013) analyzed dust samples collected from offices (n = 31), homes (n = 30), and vehicles (n = 13) in Boston, Massachusetts. PFHxA was detected in 68% of the office samples at concentrations ranging from 5.1 to 102 ng/g, 57% of the home samples at concentrations ranging from 4.9 to 1,380 ng/g, and 54% of the vehicle samples at concentrations ranging from 5.0 to 18.2 ng/g.

Water

EPA conducted monitoring for several PFAS in drinking water as part of the third and fifth Unregulated Contaminant Monitoring Rules (UCMR3 and UCMR5) (U.S. EPA, 2019b, 2016d). PFHxA was added to UCMR5 for public water system monitoring, which applies to 2022–2026 with sample collection occurring between 2023 and 2025. Some drinking water PFHxA data are available from other publications. For example, samples from seven municipal wells in Oakdale, Minnesota were analyzed for PFHxA where the concentrations ranged from <0.025 to 0.235 µg/L (U.S. EPA, 2016d). PFHxA was also detected in 23% of raw water samples collected from public water systems in New Jersey at concentrations ranging from nondetectable to 0.017 µg/L (Post et al., 2012). In a more recent study of surface waters sampled from 11 waterways in New Jersey, PFHxA was detected in 10 samples, ranging from 0.0015 to 0.026 µg/L (Goodrow et al., 2020).

AFFF Training Sites

PFHxA was detected at an Australian training ground where AFFFs had been used. Baduel et al. (2015) and Bräunig et al. (2017) observed mean concentrations of PFHxA of 0.6 µg/L in water, 8.4 µg/kg dry weight in soil, and 3.0 µg/kg wet weight in grass at an Australian town where the groundwater had been impacted by PFAS from a nearby firefighting training facility. Houtz et al. (2013) analyzed samples of groundwater, soil, and aquifer solids collected at an Air Force firefighting training facility in South Dakota where AFFF had been used. PFAS concentrations in groundwater decreased with increased distance from the burn pit, and PFHxA was detected at a median concentration of 36 µg/L. PFHxA was detected in surficial soil at a median concentration of 11 µg/kg and in aquifer solids at a median concentration of 45 µg/kg.

Military and National Priorities List (NPL) Sites

PFHxA levels in environmental samples collected in 2014 have been measured at military and National Priorities List (NPL) sites in the United States. Table 1-2 provides the concentrations at these sites (ATSDR, 2021; Anderson et al., 2016).

Table 1-2. PFHxA levels at 10 military installations and National Priority List sites

Media	PFHxA value	Site	Source
Surface soil Frequency of detection (%) Median (ppb) Maximum (ppb)	 70.33 1.75 51.0	Military[a]	Anderson et al. (2016)
Subsurface soil Frequency of detection (%) Median (ppb) Maximum (ppb)	 65.38 1.04 140	Military[a]	Anderson et al. (2016)
Sediment Frequency of detection (%) Median (ppb) Maximum (ppb)	 63.64 1.70 710	Military[a]	Anderson et al. (2016)
Surface Water Frequency of detection (%) Median (ppb) Maximum (ppb)	 96.00 0.320 292	Military[a]	Anderson et al. (2016)
Groundwater Frequency of detection (%) Median (ppb) Maximum (ppb)	 94.20 0.820 120	Military[a]	Anderson et al. (2016)
Water (ppb) Median Geometric mean	 0.25 0.10	NPL[b]	ATSDR (2021)
Soil (ppb) Median Geometric mean	 1,175 1,175	NPL[b]	ATSDR (2021)
Air (ppbv) Median Geometric mean	 ND ND	NPL[b]	ATSDR (2021)

[a]Samples collected between March and September 2014 from 10 active U.S. Air Force installations located throughout the United States, including Alaska, with a historic use of AFFFs; data originally reported as µg/kg.
[b]Concentrations found in ATSDR site documents; water and soil values represent data from two NPL sites.

Other Exposures

Schecter et al. (2012) collected 31 food samples from 5 grocery stores in Texas and analyzed them for persistent organic pollutants, including PFHxA. PFHxA was not detected in the samples. Chen et al. (2018) analyzed PFAS in a wide range of foods in Taiwan and detected PFHxA at geometric mean concentrations ranging from 0.03 ng/mL in milk to 1.58 ng/g in pork liver. Heo

et al. (2014) analyzed a variety of foods and beverages in Korea for PFAS. PFHxA was detected in 8.1% of the fish and shellfish samples at a mean concentration of 0.037 ng/g; 8.1% of the dairy samples at a mean concentration of 0.051 ng/g; 9.5% of the beverage samples at a concentration of 0.187 ng/L; 20.5% of the fruit and vegetable samples at a mean concentration of 0.039 ng/g; and 51.3% of the meat samples at a mean concentration of 0.515 ng/g. Heo et al. (2014) also detected PFHxA in tap water in Korea at a mean concentration of 11.7 ng/L; PFHxA was not detected in bottled water. Pérez et al. (2014) analyzed PFAS in 283 food items (38 from Brazil, 35 from Saudi Arabia, 36 from Serbia, and 174 from Spain). PFHxA was detected in 6.0%, 21.3%, and 13.3% of the samples from Brazil, Saudi Arabia, and Spain, respectively. The mean concentrations of PFHxA were 270, 931, and 418 pg/g sample, respectively. The study did not find PFHxA in any of the Serbian samples. PFHxA was detected in microwave popcorn packaging materials at a range of 3.4 to 497 ng/g but was not detected in the corn or popcorn (Moreta and Tena, 2014).

Stahl et al. (2014) characterized PFAS in freshwater fish from 164 U.S. urban river sites and 157 near-shore Great Lakes sites. PFHxA was not detected in the fish from U.S. urban rivers but was detected in fish from 15% of the Great Lakes sites at a maximum concentration of 0.80 ng/g.

1.2. SUMMARY OF ASSESSMENT METHODS

This section summarizes the methods used for developing this assessment. A detailed description of these methods is provided in the PFAS Systematic Review Protocol for the PFDA, PFNA, PFHxA, PFHxS, and PFBA IRIS Assessments (see Appendix A and online). The protocol includes additional problem formulation details, including the specific aims and key science issues identified for this assessment.

1.2.1. Literature Search and Screening

The detailed search approach, including the query strings and populations, exposures, comparators, and outcomes (PECO) criteria, are provided in Appendix A, Table 3-1. The results of the current literature search and screening efforts are documented in Section 2.1. Briefly, a literature search was first conducted in 2017 and regular yearly updates have been performed (the literature fully considered in the assessment was until April 2021). As described in the protocol, studies identified after peer review begins will only be considered for inclusion if they meet the PECO criteria and are expected to fundamentally alter the assessment's conclusions (see Appendix A, Section 4.1).

The literature search queries the following databases (no literature was restricted by language):

- PubMed (National Library of Medicine)
- Web of Science (Thomson Reuters)
- Toxline (moved to PubMed)

- TSCATS (Toxic Substances Control Act Test Submissions)

In addition, relevant literature not found through evidence base searching was identified by:

- Review of studies cited in any PFHxA PECO-relevant studies and published journal reviews; finalized or draft U.S. state, U.S. federal, and international assessments (e.g., the draft Agency for Toxic Substances and Disease Registry [ATSDR] assessment released publicly in 2018). In addition, studies included in ongoing IRIS PFAS assessments (PFHxS, PFNA, PFDA) were also scanned for any studies that met PFHxA PECO criteria.

- Review of studies submitted to federal regulatory agencies and brought to EPA's attention. For example, studies submitted to EPA by the manufacturers in support of requirements under the Toxic Substances Control Act (TSCA).

- Identification of studies during screening for other PFAS. For example, epidemiological studies relevant to PFHxA sometimes were identified by searches focused on one of the other four PFAS currently being assessed by the IRIS Program.

- Other gray literature (i.e., primary studies not indexed in typical evidence bases, such as technical reports from government agencies or scientific research groups; unpublished laboratory studies conducted by industry; or working reports/white papers from research groups or committees) brought to EPA's attention.

All literature, including literature search updates, is tracked in the EPA Health and Environmental Research Online (HERO) database.[3] The PECO criteria identify the evidence that addresses the specific aims of the assessment and focuses the literature screening, including study inclusion/exclusion (see Table 1-3).

Table 1-3. Populations, exposures, comparators, and outcomes (PECO) criteria

PECO element	Evidence
Populations	Human: Any population and lifestage (occupational or general population, including children and other sensitive populations). The following study designs will be included: controlled exposure, cohort, case-control, and cross-sectional. (Note: Case reports and case series will be tracked as potential supplemental material.) Animal: Nonhuman mammalian animal species (whole organism) of any lifestage (including preconception, in utero, lactation, peripubertal, and adult stages). Other: In vitro, in silico, or nonmammalian models of genotoxicity. (Note: Other in vitro, in silico, or nonmammalian models will be tracked as potential supplemental material.)
Exposures	Human: Studies providing quantitative estimates of PFAS exposure based on administered dose or concentration, biomonitoring data (e.g., urine, blood, or other specimens), environmental or occupational-setting measures (e.g., water levels or air concentrations, residential location and/or

[3]EPA's Health and Environmental Research Online (HERO) database provides access to the scientific literature behind EPA science assessments. The database includes more than 3,000,000 scientific references and data from the peer-reviewed literature EPA uses to develop its risk assessments and related regulatory decisions.

PECO element	Evidence
	duration, job title, or work title). (Note: Studies that provide qualitative, but not quantitative, estimates of exposure will be tracked as supplemental material.) Animal: Oral or Inhalation studies including quantified exposure to a PFAS of interest based on administered dose, dietary level, or concentration. (Note: Nonoral and noninhalation studies will be tracked as potential supplemental material.) PFAS mixture studies are included if they employ an experimental arm that involves exposure to a single PFAS of interest. (Note: Other PFAS mixture studies are tracked as potential supplemental material.) Studies must address exposure to one or more of the following: PFHxA (CASRN 307-24-4), PFHxA sodium salt (CASRN 2923-26-4), PFHxA ammonium salt (CASRN 21615-47-4). [Note: although while these PFHxA is not metabolized or transformed in the body, there are precursor compounds known to be biotransformed to a PFAS of interest; for example, 6:2 fluorotelomer alcohol is metabolized to PFHxA and PFBA (Russell et al., 2015). Thus, studies of precursor PFAS that identify and quantify a PFAS of interest will be tracked as potential supplemental material (e.g., for ADME analyses or interpretations).]
Comparators	Human: A comparison or reference population exposed to lower levels (or no exposure/exposure below detection levels) or for shorter periods of time. Animal: Includes comparisons to historical controls or a concurrent control group that is unexposed, exposed to vehicle-only or air-only exposures. (Note: Experiments including exposure to PFAS across different durations or exposure levels without including one of these control groups will be tracked as potential supplemental material [e.g., for evaluating key science issues; Section 2.4].)
Outcomes	All cancer and noncancer health outcomes. (Note: Other than genotoxicity studies, studies including only molecular endpoints [e.g., gene or protein changes; receptor binding or activation] or other nonphenotypic endpoints addressing the potential biological or chemical progression of events contributing toward toxic effects will be tracked as potential supplemental material [e.g., for evaluating key science issues; Section 2.4].)
PBPK models	Studies describing physiologically based pharmacokinetic (PBPK) and other PK models for PFDA (CASRN 335-76-2), PFDA ammonia salt (CASRN 3108-42-7), PFDA sodium salt (CASRN 3830-45-3), PFNA (CASRN 375-95-1), PFNA ammonium salt (CASRN 4149-60-4), PFNA sodium salt (CASRN 21049-39-8), PFHxA (CASRN 307-24-4), PFHxS (CASRN 355-46-4), PFHxS potassium salt (CASRN 3871-99-6), PFBA (CASRN 375-22-4), or PFBA ammonium salt (CASRN 10495-86-0).

ADME = absorption, distribution, metabolism, and excretion; PK = pharmacokinetic.

In addition to those studies meeting the PECO criteria and studies excluded as not relevant to the assessment, studies containing supplemental material potentially relevant to the specific aims of the assessment were inventoried during the literature screening process. Although these studies did not meet PECO criteria, they were not excluded. Rather, they were considered for use in addressing the identified key science issues (see Appendix A, Section 2.4) and other potential scientific uncertainties identified during assessment development but unanticipated at the time of protocol posting. Studies categorized as "potentially relevant supplemental material" included the following:

- In vivo mechanistic or mode-of-action studies, including non-PECO routes of exposure (e.g., intraperitoneal injection) and populations (e.g., nonmammalian models);

- In vitro and in silico models;
- Absorption, distribution, metabolism, and excretion (ADME) and pharmacokinetic (PK) studies (excluding models);[4]
- Exposure assessment or characterization (no health outcome) studies;
- Human case reports or case-series studies; and

The literature was screened by two independent reviewers with a process for conflict resolution, first at the title and abstract level and subsequently the full-text level, using structured forms in DistillerSR (Evidence Partners). Literature inventories for studies meeting PECO criteria and studies tagged as "potentially relevant supplemental material" during screening were created to facilitate subsequent review of individual studies or sets of studies by topic-specific experts.

1.2.2. Evaluation of Individual Studies

The detailed approaches used for the evaluation of epidemiological and animal toxicological studies used in the PFHxA assessment are provided in the systematic review protocol (see Appendix A, Section 6). The general approach for evaluating health effect studies meeting PECO criteria is the same for epidemiological and animal toxicological studies although the specifics of applying the approach differ. Approaches for evaluating mechanistic evidence are described in detail in Appendix A, Section 6.5. The key concerns during the review of epidemiological and animal toxicological studies are potential bias (factors that affect the magnitude or direction of an effect in either direction) and insensitivity (factors that limit the ability of a study to detect a true effect; low sensitivity is a bias toward the null when an effect exists). Briefly, for epidemiology studies, evaluation of risk of bias and study sensitivity are conducted for the following domains: exposure measurement, outcome ascertainment, participant selection, potential confounding, analysis, study sensitivity, and selective reporting. The principles and framework used for evaluating epidemiology studies are based on the Cochrane Risk of Bias in Nonrandomized Studies of Interventions (ROBINS-I) but have been modified to address environmental and occupational exposures. For animal studies, risk of bias and study sensitivity are evaluated for the following domains: reporting quality, allocation, observational bias/blinding, confounding, selective reporting and attrition, chemical administration and characterization, exposure timing, frequency and duration, endpoint sensitivity and specificity, and results presentation. Details of the human epidemiology and animal toxicology methodology, including core and prompting questions as well as PFAS-specific criteria, are described in greater detail in Appendix A (see Appendix A, Sections 6.2 and 6.3, respectively).

In evaluating individual studies, two or more reviewers independently arrived at judgments about the reliability of the study results (reflected as study confidence determinations; see below)

[4]Given the known importance of ADME data, this supplemental tagging was used as the starting point for a separate screening and review of PK data (see Appendix A, Section9.2 for details).

with regard to each outcome or outcome grouping of interest; thus, different judgments were possible for different outcomes within the same study. The results of these reviews were tracked within EPA's version of the Health Assessment Workplace Collaborative (HAWC). To develop these judgments, each reviewer assigned a rating of *good*, *adequate*, *deficient* (or *not reported*, which generally carried the same functional interpretation as *deficient*), or *critically deficient* (see Appendix A, Section 6.1 for definitions) related to each evaluation domain representing the different characteristics of the study methods that were evaluated based on the criteria outlined in HAWC. Once all domains were evaluated, the identified strengths and limitations were collectively considered by the reviewers to reach a final study confidence classification:

- *High* confidence: No notable deficiencies or concerns were identified; the potential for bias is unlikely or minimal, and the study used sensitive methodology.
- *Medium* confidence: Possible deficiencies or concerns were noted, but the limitations are unlikely have a significant impact on the results.
- *Low* confidence: Deficiencies or concerns were noted, and the potential for bias or inadequate sensitivity could have a significant impact on the study results or their interpretation. *Low* confidence results were given less weight compared to *high* or *medium* confidence results during evidence synthesis and integration (see Section 1.2.4).
- *Uninformative*: Serious flaw(s) were identified that make the study results unusable. *Uninformative* studies were not considered further, except to highlight possible research gaps.

Using the HAWC platform (and conflict resolution by an additional reviewer, as needed), the reviewers reached a consensus judgment regarding each evaluation domain and overall (confidence) determination. The specific limitations identified during study evaluation were carried forward to inform the synthesis (see Section 1.2.4) within each body of evidence for a given health effect (i.e., study confidence determinations were not used to inform judgments in isolation).

1.2.3. Data Extraction

The detailed data extraction approach is provided in Appendix A, Section 8, and data extraction and content management is carried out using HAWC (see Appendix C). Data extraction elements that may be collected from epidemiological, controlled human exposure, animal toxicological, and in vitro studies are available in HAWC. As described in the systematic review protocol (see Appendix A), not all studies that meet the PECO criteria go through data extraction: For example, studies evaluated as being *uninformative* are not used to inform assessment judgments and, therefore, do not undergo full data extraction.

All findings from informative studies are considered for extraction, regardless of statistical significance. The level of extraction for specific outcomes within a study might differ (e.g., ranging from a qualitative description to full extraction of dose response effect size information). For

quality control, data extraction is performed by one member of the evaluation team and independently verified by at least one other member. Discrepancies in data extraction are resolved by discussion or consultation with a third member of the evaluation team.

1.2.4. Evidence Synthesis and Integration

For the purposes of this assessment, evidence synthesis and integration are considered distinct but related processes (see Appendix A, Sections 9 and 10 for full details). For each assessed health effect, the evidence syntheses provide a summary discussion of each body of evidence considered in the review that directly informs the integration across evidence that was used to draw an overall judgment for each health effect. The available human and animal evidence pertaining to the potential health effects were synthesized separately, with each synthesis resulting in a summary discussion of the available evidence that addresses considerations regarding causation adapted from Hill (1965). Briefly, the following aspects are considered for the available evidence: study confidence, consistency, strength (effect magnitude) and precision, biological gradient/dose response, coherence, mechanistic evidence related to biological plausibility, and natural experiments (applicable to human studies only). Detailed descriptions and application of these considerations are described in Appendix A, Section 9. Mechanistic evidence is also synthesized as necessary to help inform key decisions regarding the human and animal evidence; processes for synthesizing mechanistic information are covered in detail in Appendix A, Section 9.2.

The syntheses of the human and animal health effects evidence focus on describing aspects of the evidence that best inform causal interpretations, including the exposure context examined in the sets of available studies. The evidence synthesis is based primarily on studies of *high* and *medium* confidence. The systematic review protocol (see Appendix A) describes that, in certain instances (i.e., few or no studies with higher confidence are available), *low* confidence studies might be used to help evaluate consistency, or if the study designs of the *low* confidence studies address notable uncertainties in the set of *high* or *medium* confidence studies on a given health effect. If *low* confidence studies are used, a careful examination of the study evaluation and sensitivity with potential effects on the evidence synthesis conclusions will be included in the narrative. For the current assessment all studies meeting PECO criteria were used for evidence synthesis and included in the narrative. When possible, results across studies are compared using graphs and charts or other data visualization strategies. The synthesis of mechanistic information informs the integration of health effects evidence for both hazard identification (e.g., biological plausibility or coherence of the available human or animal evidence; inferences regarding human relevance, or the identification of susceptible populations and lifestages across the human and animal evidence) and dose-response evaluation (e.g., selection of benchmark response levels, selection of uncertainty factors). Evaluations of mechanistic information typically differ from evaluations of phenotypic evidence (e.g., from routine toxicological studies). This is primarily because mechanistic data evaluations consider the support for and involvement of specific events or sets of events within the context of a broader research question (e.g., support for a hypothesized mode of action; consistency

with known biological processes), rather than evaluations of individual apical endpoints considered in relative isolation.

Following the synthesis of human and animal health effects data, and mechanistic data, integrated judgments are drawn across all lines of evidence for each assessed health effect. During evidence integration, a structured and documented process was used, as follows:

- Building from the separate syntheses of the human and animal evidence, the strength of the evidence from the available human and animal health effect studies was summarized in parallel, but separately, using a structured evaluation of an adapted set of considerations first introduced by Bradford Hill (Hill, 1965). This process is similar to that used by the Grading of Recommendations Assessment, Development, and Evaluation (GRADE) (Morgan et al., 2016; Guyatt et al., 2011; Schünemann et al., 2011), which arrives at an overall integration conclusion based on consideration of the body of evidence. These summaries incorporate the relevant mechanistic evidence (or mode of action [MOA] understanding) that informs the biological plausibility and coherence within the available human or animal health effect studies. The terms associated with the different strength of evidence judgments within evidence streams are *robust*, *moderate*, *slight*, *indeterminate*, and *compelling evidence of no effect*.

- The animal, human, and mechanistic evidence judgments are then combined to draw an overall judgment that incorporates inferences across evidence streams. Specifically, the inferences considered during this integration include the human relevance of the animal and mechanistic evidence, coherence across the separate bodies of evidence, and other important information (e.g., judgments regarding susceptibility). Note that without evidence to the contrary, the human relevance of animal findings is assumed. The final output is a summary judgment of the evidence base for each potential human health effect across evidence streams. The terms associated with these summary judgments are *evidence demonstrates*, *evidence indicates (likely)*, *evidence suggests*, *evidence inadequate*, and *strong evidence of no effect*. The decision points within the structured evidence integration process are summarized in an evidence profile table for each considered health effect.

- In some instances (i.e., key science questions, coherence within and across biologically related outcomes, areas of uncertainty) the supplemental mechanistic information was reviewed and prioritized based on potential impact on assessment conclusions. For example, interpreting a pattern of changes or collection of findings from various health outcomes is strengthened by biological understanding (e.g., disruption of thyroid homeostasis through increased clearance of thyroid hormones during gestation may affect fetal growth and nervous system development) and progression to adverse outcomes and applicability in humans. Biological understanding, including strong mechanistic support for the chemical molecular interactions and conservation of those reactions and subsequent biological responses between species can increase certainty in strength of the evidence.

As discussed in the protocol (see Appendix A), the methods for evaluating the potential carcinogenicity of PFAS follow processes laid out in the EPA cancer guidelines (U.S. EPA, 2005) and that the judgments described here for different cancer types are used to inform the evidence integration narrative for carcinogenicity and selection of one of EPA's standardized cancer descriptions. These are: (1) *carcinogenic to humans*, (2) *likely to be carcinogenic to humans*, (3)

suggestive evidence of carcinogenic potential, (4) *inadequate information to assess carcinogenic potential*, or (5) *not likely to be carcinogenic to humans*. However, for PFHxA, data relevant to cancer were sparse and did not allow for such an evaluation (see Section 3.3).

1.2.5. Dose-Response Analysis

The details for the dose-response analysis completed for this assessment are in Appendix A, Section 11. Briefly, although procedures for dose-response assessments were developed for both noncancer and cancer health hazards, and for the oral route of exposure following exposure to PFHxA, the existing data for PFHxA only supported derivation of an oral reference dose (RfD) for noncancer hazards (see Appendix A, Section 11 for the health hazard conclusions necessary for deriving other values). An RfD is an estimate, with uncertainty spanning perhaps an order of magnitude, of an exposure to the human population (including susceptible subgroups) that is likely without an appreciable risk of deleterious health effects over a lifetime (U.S. EPA, 2002c). The derivation of a reference value like the RfD depends on the nature of the health hazard conclusions drawn during evidence integration. For noncancer outcomes, dose-response assessments were conducted for evidence integration judgments of ***evidence demonstrates*** and ***evidence indicates (likely)***. In general, toxicity values are not developed for noncancer hazards with *evidence suggests* conclusions (see Appendix A, Section 10.2 for exceptions).
Consistent with EPA practice, the PFHxA assessment applied a two-step approach for dose-response assessment that distinguishes analysis of the dose-response data in the range of observation from any inferences about responses at lower, environmentally relevant exposure levels (U.S. EPA, 2012a, 2005).

- Within the observed dose range, the preferred approach is to use dose-response modeling to incorporate as much of the data set as possible into the analysis, and considering guidance on modeling dose-response data, assessing model fit, selecting suitable models, and reporting modeling results [see the EPA Benchmark Dose Technical Guidance (U.S. EPA, 2012a)] as elaborated in Appendix A, Section 11. Thus, modeling to derive a POD attempted to include an exposure level near the lower end of the range of observation, without significant extrapolation to lower exposure levels.

- As derivation of cancer risk estimates and reference values nearly always involves extrapolation to exposures lower than the POD; the approaches to be applied in these assessments are described in more detail in Appendix A, Section 11.2.

When sufficient and appropriate human and laboratory animal data are available for the same outcome, human data are generally preferred for the dose-response assessment because use of human data eliminates the need to perform interspecies extrapolations. For reference values, this assessment will derive a candidate value from each suitable data set. Evaluation of these candidate values grouped within a given organ/system were used to derive a single organ/system-specific RfD (osRfD) for each organ/system under consideration from which a single overall reference value will be selected to cover all health outcomes across all organs/systems.

Although this overall RfD represents the focus of the dose-response assessment, the osRfDs can be useful for subsequent cumulative risk assessments that consider the combined effect of multiple PFAS (or other agents) acting at a common organ/system. For noncancer toxicity values, uncertainties in these estimates are characterized and discussed. In addition, a less-than-lifetime, "subchronic" RfD was similarly estimated. Uncertainties in these toxicity values are transparently characterized and discussed. For dose-response purposes, EPA has developed a standard set of models (http://www.epa.gov/bmds) that can be applied to typical data sets, including those that are nonlinear. In situations where alternative models with significant biological support are available (e.g., pharmacodynamic models), those models are included as alternatives in the assessment(s) along with a discussion of the models' strengths and uncertainties. EPA has developed guidance on modeling dose-response data, assessing model fit, selecting suitable models, and reporting modeling results [see the EPA Benchmark Dose Technical Guidance (U.S. EPA, 2012a)]. Additional judgment or alternative analyses are used if the procedure fails to yield reliable results; for example, if the fit is poor, modeling might be restricted to the lower doses, especially if competing toxicity at higher doses occurs. When alternative approaches fail or are not applicable, the NOAEL/LOAEL approach is used for POD estimation. For each modeled response, a POD from the observed data was estimated to mark the beginning of extrapolation to lower doses. The POD is an estimated dose (expressed in human-equivalent terms) near the lower end of the observed range without significant extrapolation to lower doses. The POD is used as the starting point for subsequent extrapolations and analyses. For noncancer effects, the POD is used in calculating the RfD.

2. SUMMARY OF LITERATURE IDENTIFICATION AND STUDY EVALUATION RESULTS

2.1. LITERATURE SEARCH AND SCREENING RESULTS

The evidence base searches yielded 975 unique records identified from core database searches (WOS, Toxline, PubMed, Toxnet) and other sources (see Figure 2-1). Of the 975 studies identified, 74 were excluded at the title and abstract level on the basis that they did not meet the PECO and did not contain potentially relevant supplemental information. At the full-text level, 57 studies met PECO criteria including 24 human health effect studies, 23 in vivo animal studies, 7 genotoxicity study, and 5 PBPK/PK studies (see Table 1-3). An interactive summary of the literature screening results is available as a literature tag-tree accessible via the link: PFHxA Literature Tagtree. Additional information, including high-throughput screening data on PFHxA, are available from EPA's CompTox Chemicals Dashboard (U.S. EPA, 2018a).

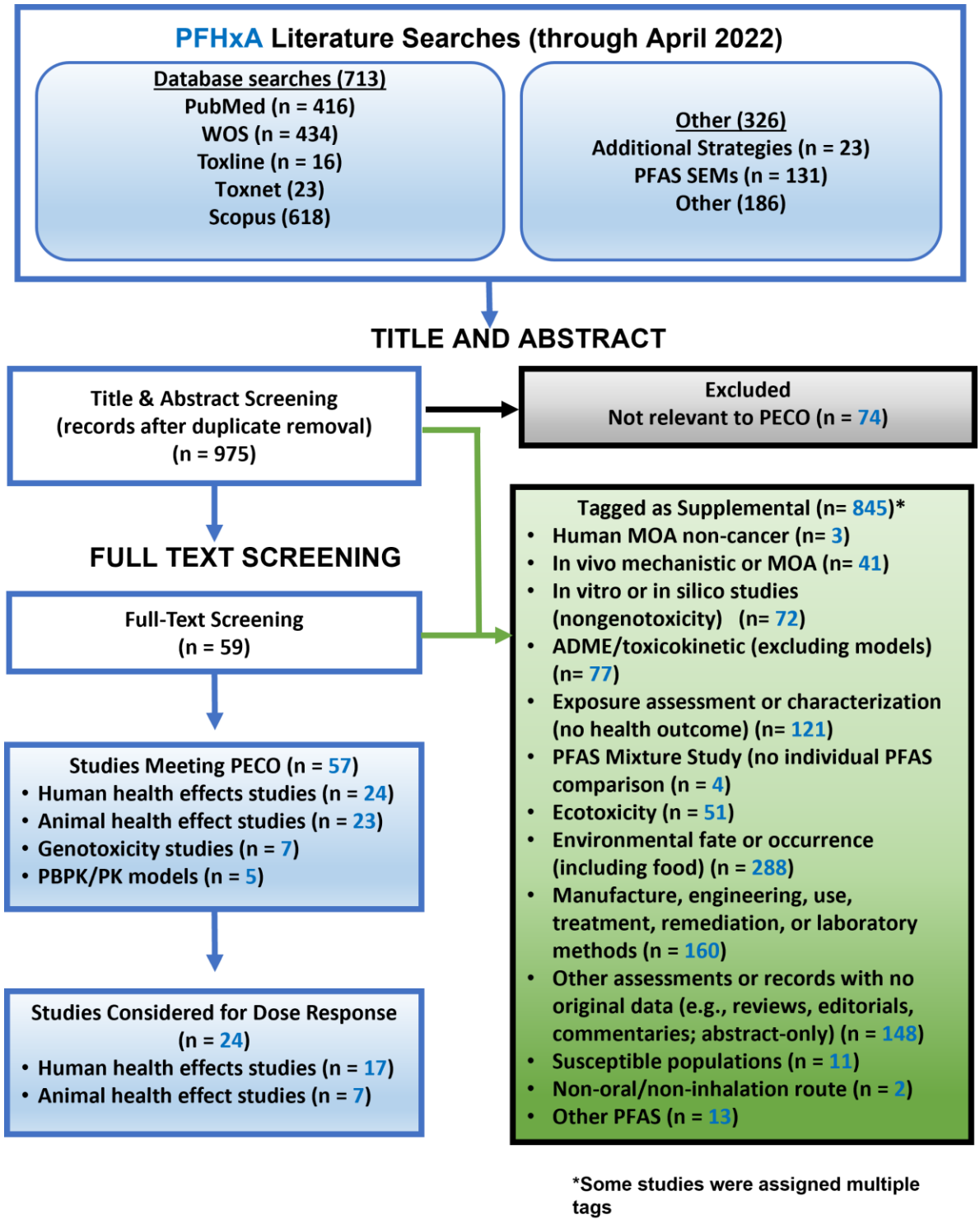


Figure 2-1. Literature search and screening flow diagram for PFHxA and related salts, PFHxA-NH_4 and PFHxA-Na. Literature HAWC tree are available in HAWC and all studies are available in PFHxA HERO.

PFHxA-NH_4 = ammonium perfluorohexanoate; PFHxA-Na = sodium perfluorohexanoate.

2.2. STUDY EVALUATION RESULTS

Human and animal studies evaluated potential hepatic, developmental, hematopoietic, endocrine, cardiometabolic, renal, reproductive, immune, and nervous system effects following exposure to PFHxA. The evidence informing these potential health effects is presented and assessed in Sections 3.2.1–3.2.9. Seventeen epidemiological studies were identified that report on the potential association between PFHxA and human health effects. Of these, five were considered *uninformative* due to critical deficiencies in one or more domains, including participant selection, exposure measurement, confounding, or analysis (Zhang et al., 2019; Seo et al., 2018; Kim et al., 2016a; Jiang et al., 2014). The remaining 12 studies were rated *medium* (Liu et al., 2022; Nian et al., 2019; Bao et al., 2017; Zeng et al., 2015; Dong et al., 2013) or *low* confidence (Velarde et al., 2022; Tian et al., 2019; Wang et al., 2019; Song et al., 2018; Li et al., 2017; Zhou et al., 2016; Fu et al., 2014).

Seven unique reports of animal studies met PECO criteria. The available evidence base of animal toxicity studies on PFHxA and the related ammonium and sodium salts primarily consists of five reports in rats and mice including short-term (NTP, 2018), subchronic (Chengelis et al., 2009b; Loveless et al., 2009), chronic (Klaunig et al., 2015), and reproductive/developmental (Iwai and Hoberman, 2014; Loveless et al., 2009) experiments. These studies were generally well conducted and judged *high* or *medium* confidence. In cases where a study was rated *low* confidence for one or more of the evaluated outcomes, the specific limitations identified during evaluation are discussed in the applicable synthesis section(s). An acute (<24 hours) oral toxicity study was also available (Riker Labs, 1979) but not considered informative as other longer duration (>30 day) studies were available. A reproductive study by Kirkpatrick (2005a) was initially considered *uninformative* due to reporting deficiencies (i.e., all summary and individual animal data [pages 110–1,334] were missing). During public comment, the study sponsor submitted the missing data tables (Kirkpatrick, 2005a, b, c, d, e, f, g). Consistent with the protocol and the study was reconsidered for incorporation in the assessment based on the potential impact on assessment conclusions. The study was considered *medium confidence* for most outcomes. Immune findings from this study were incorporated into the evidence synthesis on the basis that, although they do not change the assessment conclusions, they may help to address critical data gaps for this potential health effect. For all other health effects it was determined that the data would not impact assessment conclusions based on the following rationales: for reproductive, developmental and renal effects, the findings were weak or null and did not address key data gaps; for hepatic and hematopoietic effects, the results were similar to effects observed in other studies and supported hematopoietic and hepatic conclusions (i.e., decreased hemoglobin, MCH, MCHC, total protein, globulin; increased relative liver weight, incidence of hepatocellular hypertrophy), but largely limited to the high dose that was associated with overt toxicity, including high mortality (approx. 33%) which may impact the reliability of the results. Thus, these findings were not incorporated into the evidence synthesis

and integration sections for hepatic, hematopoietic, developmental, renal, and reproductive health effects.

Detailed rationales for each domain and overall confidence rating are available in HAWC. Results shown below in Figure 2-2 for human studies are (interactive version available: link) and Figure 2-3 for animal studies (interactive version available: link). Graphical representations of the outcome-specific ratings are also presented in the organ/system-specific integration sections (in Section 3.2). All outcomes rated *low* confidence or higher were used for evidence synthesis and integration.

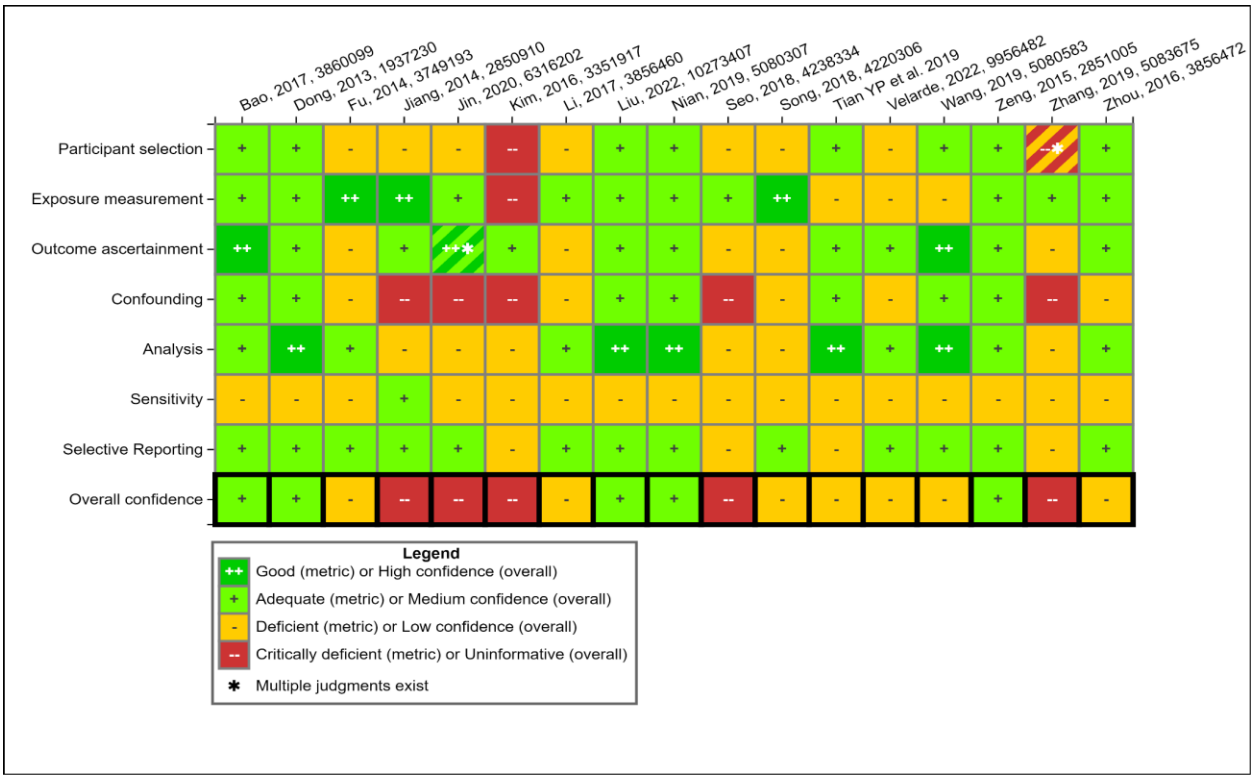


Figure 2-2. Study evaluation results for human epidemiology studies.

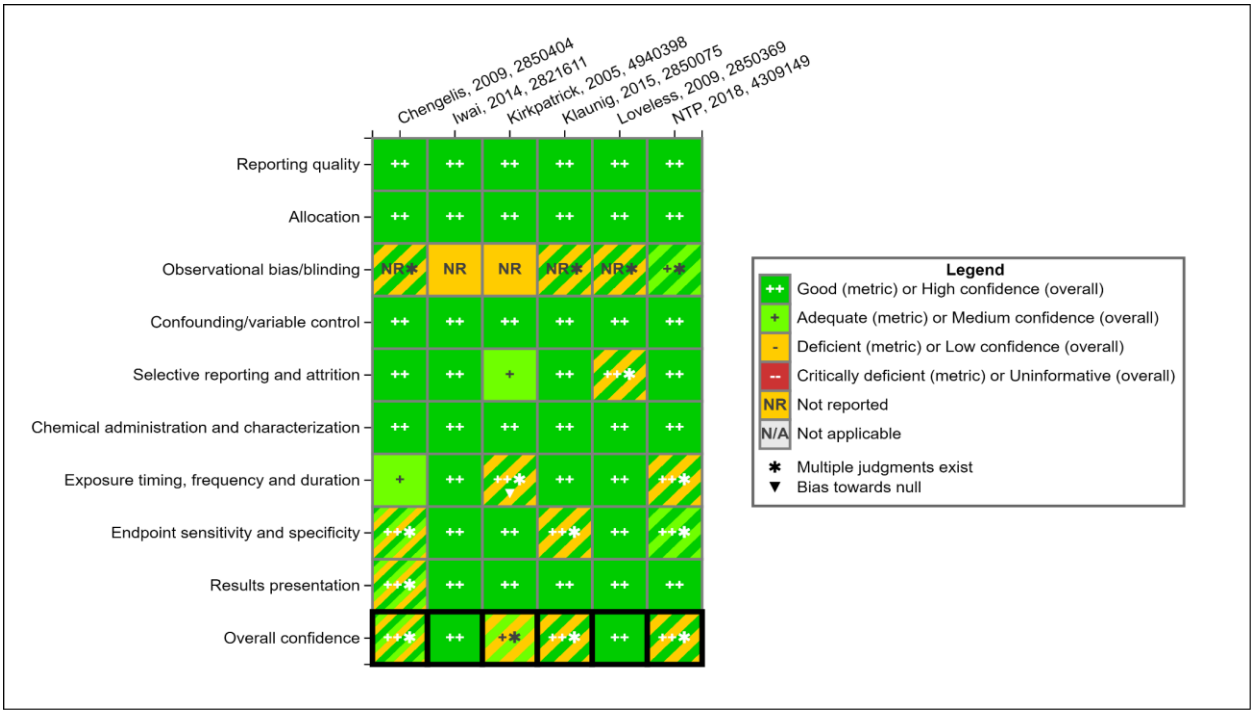


Figure 2-3.Study evaluation results for animal toxicology studies.

3. PHARMACOKINETICS, EVIDENCE SYNTHESIS, AND EVIDENCE INTEGRATION

3.1. PHARMACOKINETICS

Only a few human PK studies on PFHxA are available, but the studies provide sufficient data to estimate PFHxA half-life, a dependent variable for the estimation of clearance (along with volume of distribution). Several studies such as Ericson et al. (2007) reported PFHxA in blood or serum of human populations (e.g., in relation to age and sex) but, because exposure levels are not known for the subjects and the concentrations are not measured over time in specific subjects for whom the exposure level is known to be zero, such observations cannot be used to obtain ADME information. Several other studies that investigate specific aspects of PFHxA ADME in humans are discussed briefly below but were not used in the derivation of toxicity values. One analysis provides an estimate of PFHxA elimination in humans (Russell et al., 2013) using data from an observational study by Nilsson et al. (2013). Luz et al. (2019) describes a reanalysis of these data but based only on the three participants with the most rapid clearance. While EPA considers the data reported by Nilsson et al. (2013) to be sufficient for the estimation of a half-life in humans, the approaches used by Russell et al. (2013) and Luz et al. (2019) were not considered adequate. Therefore, the data of Nilsson et al. (2013) have been re-analyzed as described in Approach for Animal-Human Extrapolation of PFHxA Dosimetry (see Section 5.2.1).

Animal experiments in rats, mice, and monkeys have provided valuable information on PK processes of PFHxA. In brief, PFHxA and other perfluoroalkyl acids (PFAA) have similar PK aspects: They are well absorbed following oral exposure and quickly distribute throughout the body (Iwabuchi et al., 2017), particularly to blood, liver, skin, and kidney (Gannon et al., 2011). Dzierlenga et al. (2019) noted that following intravenous (i.v.) administration of 40 mg/kg PFHxA, the PK profiles were generally similar between sexes, but a lower dose-normalized area under the curve (AUC, 3.05 mM·h/mmol/kg), a faster clearance (CL, 327 mL/h-kg), and a lower volume of distribution of peripheral compartment (V_2 = 59.6 mL/kg) was observed in female Sprague-Dawley rats, as compared to their male counterparts (dose-normalized AUC = 7.38 mM·h/(mmol/kg), CL = 136 mL/h-kg, and V_2 = 271 mL/kg, respectively). Likewise, kinetic parameters (e.g., the maximum concentration [C_{max}]) were comparable between sexes following an oral dose of 40 mg/kg, except that female rats exhibited a lower dose-adjusted AUC/dose and a faster CL. A PK study in mice similarly showed an AUC/dose in male animals 2–3 times higher than in females, indicating slower elimination in males (Gannon et al., 2011). Thus, apparent sex-related quantitative differences in PFHxA PK occur in rats and mice. On the other hand, the AUC in monkeys

given a 10 mg/kg i.v. dose of PFHxA was only slightly lower in females than in males (75 vs. 84 mg-h/L), suggesting no significant sex difference in nonhuman primates.

PFHxA is resistant to metabolic transformation, and urinary excretion is the main elimination route, followed by feces (Gannon et al., 2011; Iwai, 2011; Chengelis et al., 2009a).

3.1.1. Absorption

Absorption is rapid in rodents and monkeys (Iwabuchi et al., 2017; Gannon et al., 2011; Chengelis et al., 2009a). PFHxA was extensively absorbed with an average time to reach maximum concentration ($T_{max)}$ of 1 hour in Sprague-Dawley rats given 26-day repeated gavage doses of 50, 150 or 300 mg PFHxA/kg (Chengelis et al., 2009a). After gavage at 2 or 100 mg [1-^{14}C]PFHxA/kg using a single dose or 14 daily consecutive doses, Gannon et al. (2011) also observed a short T_{max} of 30 and 15 minutes, respectively, in male and female Sprague-Dawley rats. Similarly, rapid absorption was also observed in CD-1 mice (Gannon et al., 2011). For female rats and male and female mice, PFHxA absorption does not appear to be saturated between 2 and 100 mg/kg as suggested by dose normalized $AUC_{0\rightarrow168}$ hour, but the data in male rats indicate either a 25% reduction in absorption or a corresponding increase in PFHxA clearance between these two dose levels (Gannon et al., 2011; Chengelis et al., 2009a).

In a recent PK study by Dzierlenga et al. (2019), Sprague-Dawley rats were given PFHxA, by i.v. injection (40 mg/kg) or gavage (40, 80, and 160 mg/kg). Besides collection of blood samples to evaluate the time course of plasma PFHxA for each dose and route, liver, kidney, and brain samples were collected to determine the distributions of PFHxA in tissues following 80 mg/kg gavage dose. A two-compartmental model was used to evaluate the PK profiles. The estimated oral bioavailability for PFHxA was >100% (Dzierlenga et al., 2019); this result simply could reflect experimental and analytical uncertainty in estimating the serum concentration AUC from intravenous versus oral exposure, but also might be due to increased reabsorption from the intestinal lumen by intestinal transporters of material excreted in the bile. The data indicate that T_{max} increased slightly but not significantly with increasing oral PFHxA dose levels for both sexes. For instance, T_{max} increased from 0.668 ± 0.154 to 0.890 ± 0.134 hour (mean ± standard error) and from 0.529 ± 0.184 to 0.695 ± 0.14 hour with increased gavage doses of PFHxA for male and female rats, respectively (Dzierlenga et al., 2019).

3.1.2. Distribution

PFHxA has an aqueous solubility of 15.7 g/L (Zhou et al., 2010). Computational chemistry predictions conclude that PFHxA and its salts have a $pK_a \leq 0$ (Rayne and Forest, 2010), so it likely exists exclusively in anionic form at physiological pH (Russell et al., 2013). Therefore, it is relatively water soluble, but limited data are available to examine its distribution to various organs and tissues upon exposure in mammalian systems (Russell et al., 2013; Gannon et al., 2011). The largest concentrations were found in liver, skin, heart, lung, and kidney and concentrations peaked within hours (Iwabuchi et al., 2017; Gannon et al., 2011). For example, Gannon et al. (2011) reported

heart, kidneys, liver, and lungs had detectable but not quantifiable concentrations of PFHxA at 24 hours in rats dosed with 100 mg/kg (Gannon et al., 2011). Similarly, the highest uptake concentrations occurred in the liver and femur (10 ± 2% and 5 ± 1% of the injected dose, respectively), in male CD-1 mice (Burkemper et al., 2017). As described in detail below, the volume of distribution (V_d) was generally similar (within a factor of three) among male and female mice, rats, and monkeys (Russell et al., 2013).

Distribution in Animal (Rats, Mice, and Monkeys) and In Vitro Studies

Chengelis et al. (2009a) gave both Sprague Dawley rats and cynomolgus monkeys (3/sex) PFHxA (10 mg/kg) via a single i.v. injection to determine PFHxA PK using noncompartmental analysis. In monkeys they observed a distribution phase of 8 hours and an apparent V_d of 0.77 and 0.35 L/kg in males and females, respectively. In male and female rats, V_d was reported as 0.18 and 0.47 L/kg, respectively, and the distribution phase after gavage dosing was about 1–2 hours in both sexes. Serum concentrations of PFHxA were up to 17-fold higher for male than female rats after i.v. dosing. In a separate experiment male and female Sprague-Dawley rats were given oral gavage doses of 50, 150, or 300 mg/kg/d PFHxA for 25 days (6 rats/sex/dose) and the PK evaluated on the first and last day of dosing. The AUC after oral dosing was approximately 4fold higher in males than females given a 50 mg/kg gavage dose on both day 1 and day 25. The half-life in males, however, was only 2.5 times greater than females after i.v. dosing and was similar to that in females after oral dosing. Together these lead to the conclusion of higher V_d for females than for males.

Using a one-compartment model, Iwabuchi et al. (2017) evaluated the distribution of PFHxA and other PFAAs (PFOA, PFOS and perfluorononanoic acid, [PFNA]) in multiple tissues (brain, heart, liver, spleen, kidney, whole blood, and serum) in 6-week-old male Wistar rats. The rats were given a single oral dose or 1- and 3-month exposures in drinking water. For the single oral dose, rats were given drinking water containing a mixture of PFAAs by gavage (PFHxA, PFOA, PFOS: 100 µg/kg body weight [BW], PFNA: 50 µg/kg BW). Although the estimated T_{max} for PFHxA was 1 hour for all tissues, the T_{max} for other PFAAs was 12 hours in the tissues except the brain (72 h) and whole blood (24 h), indicating PFHxA was distributed rapidly throughout the body. Peak concentrations occurred between 15 minutes and 1 hour after dosing, depending on the tissue. Of examined tissues, the highest concentrations of PFHxA were found in the serum and kidney, equivalent to 7.9% and 7.1% of the administered PFHxA, respectively. Note that the peak concentrations measured in liver and brain were roughly 40% (at 15 minutes) and 1.5% (at 1 hour) of the corresponding peak serum levels (4.6% and 0.027% of administered PFHxA dose), respectively. The earlier peak in liver concentration is likely due to initial delivery there from oral absorption, although the results show low delivery to the brain.

Dzierlenga et al. (2019) measured levels of PFHxA in rat liver, kidney and brain over 12 hours following an 80 mg/kg oral gavage dose. In general tissue distribution was rapid, with peak concentration occurring at 0.5 hours (first time-point) in male rat liver and kidney or 1 hour (second time-point) in male rat brain and in female rat liver, kidney, and brain. The concentrations

declined exponentially after the peak, with tissue: plasma ratios mostly remaining in a limited range. For example, in male and female rat kidney and female rat liver the tissue: plasma ratio only varied between 0.5 and 0.75, though the liver: plasma ratio varied between 1 and 0.5 in male rats, though without a clear pattern. However, the kidney: plasma ratio in female rats showed a steady increase from around 0.8 at 0.5 hours to around 1.7 at 3 hours, after which it slowly declined to around 1.4 at 12 hours (Dzierlenga et al., 2019). Since tissue: plasma ratios are generally less than 1, this result in the female rat kidney indicates a mechanism that was not active in the liver or male rats, perhaps involving active transport into the tissue.

For the 1- or 3-month exposures, rats were given a mixture of four PFAA dose levels: 0, 1, 5 and 25 μg/L in drinking water with similar intake rate across dose groups (0.072–0.077 L/kg BW-day) (Iwabuchi et al., 2017). In general, the long-term tissue concentrations of PFHxA predicted based on the data from the single-exposure studies were comparable to that measured after the 1- and 3-month exposures, suggesting that steady-state tissue levels were achieved rather quickly and the tissue distribution of PFHxA remained relatively constant over time (Iwabuchi et al., 2017).

An in vitro study using lung epithelial cells (NCI-H292) and adipocytes (3T3-L1K) made similar observations of no appreciable cellular accumulation and retention of PFHxA (Sanchez Garcia et al., 2018).

Distribution in Humans

The tissue distribution of PFHxA and other PFAAs were analyzed in 99 human autopsy samples (brain, liver, lung, bone, and kidney) (Pérez et al., 2013). Pérez et al. (2013) used the term "accumulation," which in PK terminology describes a steady increase in the amount of a substance in the body tissues over an extended time while exposure continues at a relatively constant level. So, to demonstrate accumulation, one must have repeated measures of the blood or tissue concentration in an individual over a significant period of time. If the body quickly reaches a constant level (with ongoing exposure), that would not be called "accumulation." Because the study data were collected from cadavers, they show only the tissue levels in the individuals at time of death, and thus do not actually demonstrate accumulation but simply that exposure, absorption, and distribution have occurred. These tissue concentrations could represent approximate steady-state concentrations that were achieved quickly after the start of exposure, without accumulation. More generally, these data cannot inform the specific exposure scenarios that might have occurred before the time of death, in particular the duration of exposure that was required to reach the observed concentrations.

Pérez et al. (2013) found PFHxA to be the main PFAA compound in the brain (mean = 180 ng/g tissue weight, median = 141 ng/g). PFHxA was detected in all collected tissue types at levels ranging from below the detection limit to an observed concentration of 569 ng/g in the lung. These observations generally demonstrate the *distribution* of short-chain PFAAs like PFHxA, for which the mean (or median) concentration ranged from 5.6 ng/g (2.7 ng/g) tissue in the

kidney to 180 ng/g (141 ng/g) in the brain. The liver and lung had tissue levels somewhat below that in the brain but within the same range, with mean (or median) levels of 115 ng/g (68.3 ng/g) and 50.1 ng/g (207 ng/g), respectively.

Considering the relatively rapid elimination of PFHxA in humans, the high levels reported by Pérez et al. (2013) for brain, liver and lung are surprising. Abraham et al. (2021) found much lower levels of PFBA in human tissues than Pérez et al. (2013) and attributed the discrepancy to issues in the analytic chemistry. Sanan and Magnuson (2020) describe analytic issues that can impact a broad range of PFAS, including PFHxA. Multiple PFAS may co-elute from chromatographic separation, so the measurements attributed to PFHxA by Pérez et al. (2013) may reflect multiple PFAS, with PFHxA being only a small fraction of the total. Resolution of this issue is beyond the scope of this review. Pérez et al. (2013) describes what appear to be good quality control methods and their results are reported here for completeness but are not used in the quantitative dosimetric analyses presented later.

Because blood plasma concentrations could not be evaluated in the cadavers, the data of Pérez et al. (2013) lack this component of total PFHxA body burden. Plasma is a small fraction of total body mass (~ 4% in humans), but due to PFHxA's substantial binding to serum proteins (>99% bound to serum albumin (Bischel et al., 2011)) it will carry a disproportionate amount of the PFHxA. For example, if the overall volume of distribution in humans is 0.5 L/kg, plasma will then contain about 8% of the PFHxA.

Fàbrega et al. (2015) attempted to estimate tissue: blood partition coefficients (PCs) for PFHxA using the data of Pérez et al. (2013). Because Pérez et al. (2013) did not measure or report blood concentrations, Fàbrega et al. (2015) used the mean blood concentration reported 4 years earlier for residents of the same county (Ericson et al., 2007). The resulting set of PCs ranged from 6 (unitless ratio) in the kidney to 202 in the brain, indicating a V_d in the human body around 40 L/kg or higher.

Zhang et al. (2013a) evaluated the distribution of several PFAS including PFHxA in matched samples of maternal blood, cord blood, placenta, and amniotic fluid among Chinese women. Only 45% of maternal blood samples were above the limit of quantitation (LOQ), with a mean concentration of 0.07 ng/mL, although 87% of cord blood samples were above the LOQ, with a mean of 0.21 ng/mL PFHxA. Only 17% of placenta samples were above the LOQ (mean concentration 0.04 ng/mL) and 45% of amniotic fluid samples (mean concentration 0.19 ng/mL). The authors urge caution in interpreting their results because recovery of PFHxA from test samples was more variable than for most other PFAAs. These data do show, however, that PFHxA distributes into the fetus during pregnancy.

More recently, Li et al. (2020) evaluated the transplacental transfer of multiple PFAS, including PFHxA, in preterm versus full-term births, and evaluated for correlation with the expression of nine placental transporters. The transplacental transfer efficiency (TTE) was calculated as the ratio of PFAS concentration in cord serum, collected at the time of birth, to the

concentration in maternal serum collected within 1 week of (prior to) birth. The median TTE for preterm births was 0.8, with first and third quartiles of 0.5 and 1.17, respectively (n = 27), hence the distribution in the preterm fetus was slightly less than but not significantly different from 1. This contrasts with PFDA, for example, for which the median TTE was 0.23 and the third quartile 0.29, indicating that the placenta limited distribution of PFDA to the fetus much more so than PFHxA. In full-term births the median TTE for PFHxA was 2.26 (Q1 = 0.88, Q3 = 5.30, n = 88), which is exceptionally high compared to most other PFAS for which the median TTE ranged between 0.35 (PFDA) and 1.32. Only PFTeDA had a median full-term TTE nearly as high as PFHxA, i.e., 1.84. The authors conjectured that the general shift to higher TTE in full-term pregnancies may be due to loss of placental integrity as a barrier (Li et al., 2020), but for most PFAS this is a shift from a median value below one to a value that is higher but still less than or close to one. In the case of PFHxA these results do not indicate a significant placental barrier for the preterm fetus and an elevated transfer at full term. The TTE was not significantly correlated with any of the transporters analyzed. The *p*-value for association with equilibrative nucleoside transporter (ENT1) was 0.054, indicating some possibility for its role, but with a negative correlation coefficient. There was a non-significant but positive correlation between the TTE and p-glycoprotein (MDR1), which had a significant positive association with other PFAS (Li et al., 2020). So, there is no clear evidence or explanation for why PFHxA would be elevated in full-term cord blood compared to maternal blood, but it seems possible that active transport could play a role in some individuals.

The partitioning of PFHxA and 15 perfluoroalkyl substances (C6–C11) between plasma and blood cells was investigated using blood samples collected from human subjects (n = 60) (Jin et al., 2016). The results showed that although the estimated mass fraction in plasma generally increased with the carbon chain length, PFHxA appeared to have lowest mass fraction in plasma (0.24) as compared with other PFAA chemicals (0.49 to 0.95). In a study population of 61 adults in Norway, Poothong et al. (2017) also found that although PFHxA was detected in 100% of the whole blood samples, it was not detected in serum or plasma. Given the strong partitioning to whole blood (perhaps due to partitioning into blood cells), the whole blood, rather than serum or plasma, was suggested as a better blood matrix for assessing PFHxA exposure (Poothong et al., 2017).

Synthesis of Distribution Across Species

In contrast to the estimated PCs of Fàbrega et al. (2015), Chengelis et al. (2009a) estimated V_d of 0.18 and 0.47 L/kg, respectively, in male and in female rats. For monkeys, the individual estimates of V_d Chengelis et al. (2009a) reported varied widely for each sex; for example, the coefficient of variation among the three females was 74%. Therefore, EPA recalculated male and female values for this analysis from the mean values of $AUC_{0-\infty}$ and the beta-phase elimination constant, K_{el}:

$$V_d = \text{dose}/[\text{mean}(AUC_{0-\infty}) \times \text{mean}(K_{el})]. \quad (3\text{-}1)$$

The resulting values of V_d were 0.77 L/kg and 0.35 L/kg for male and female monkeys, respectively. Although the reported values for rats and these re-estimated values for monkeys were within similar ranges, spanning less than a factor of five, the difference between males and females of each species is larger than expected. The underlying data indicate significant PK differences between males and females of each species.

The average V_d for rats (0.33 L/kg) is only 40% lower than the average for monkeys (0.56 L/kg), a modest species difference that could occur due to differences in the relative concentration of binding proteins and phospholipids in blood (e.g., albumin) versus the rest of body (Sanchez Garcia et al., 2018). Partitioning or distribution is primarily a function of the physicochemical properties of a tissue versus blood (binding site content and phospholipid concentration being significant components for PFAS) and are typically similar across mammalian species, not differing by orders of magnitude as suggested by the difference between the results of Fàbrega et al. (2015) for humans and the animal PC data. This raises a significant question about reasons for the apparent disparity. EPA is unaware of a specific mechanism that could explain this discrepancy, particularly one that differs between monkeys and humans to such a large extent but not between monkeys and rats.

Therefore, the most likely explanation for the differences in the PCs estimated by Fàbrega et al. (2015) are an artifact of combining data from nonmatched human samples Pérez et al. (2013) whereas Ericson et al. (2007) collected data over several years (e.g., due to a change in PFHxA exposure in that population across those times). Thus, these results are considered too uncertain for further analysis of human pharmacokinetics. Instead, the V_d estimated for male and female monkeys by Chengelis et al. (2009a) is assumed to provide the best estimates for men and women, respectively, given the biochemical properties of tissues that determine the relative affinity for PFHxA in tissue versus blood are more similar between humans and a nonhuman primate than between humans and rats or mice. Because the V_d in monkeys is similar to that in rats (see details above, Distribution in Animals) and an assumption of similar partitioning in humans versus other mammals has been successfully used for many PBPK models, this assumption is considered modest with minimal associated uncertainty.

A generally accepted assumption in pharmacokinetics is that renal clearance (via glomerular filtration) is limited to the fraction unbound in plasma (Janků, 1993). PFAS accumulation in tissues appears to correlate with phospholipid binding and content and like lipids the relative distribution of phospholipids, albumin, and other binding sites is not expected to differ by orders of magnitude between humans and other animals (Sanchez Garcia et al., 2018). Some evidence suggests plasma protein binding (e.g., serum albumin) could also play a role in PFHxA pharmacokinetics. A study by D'eon et al. (2010) evaluated the molecular interactions of PFHxA and PFOA with human serum albumin (HSA) using nuclear magnetic resonance spectroscopy. They found the interaction of both PFHxA and PFOA with HSA—assessed based on data for selected HSA ligands including oleic acid, phenylbutazone, and ibuprofen—could affect its pharmacokinetics.

Organic anion transporters, a family of transmembrane proteins, had been suggested to play a role in the renal reabsorption of PFAAs (Kudo, 2015; Weaver et al., 2010) (see further discussion below for rat studies). Weaver et al. (2010) found that renal transport of PFAAs with different chain lengths (C2–C18) could occur via specific transporters (Oat1, Oat2, Oat3, Urat1, and Oatp1a1) that were differentially located in the basolateral membrane and apical membrane in rats (Chinese hamster ovary cell line and kidney RNA from Sprague-Dawley rats). Although PFHxA was capable of inhibiting Oat1-mediated transport of *p*-aminohippurate, the model substrate used for PFAA transport tests, the quantitative role of organic anion transporters in PFHxA PK remains uncertain due to the rapid elimination kinetics of PFHxA (Weaver et al., 2010). The role of Oatp1a1 and its regulation by sex hormones is discussed at further length below (Rat Studies).

On the other hand, although Bischel et al. (2011) measured the binding of PFHxA to bovine serum albumin (BSA) in vitro, the measured fraction bound is 99%, which appears quantitatively inconsistent with the empirical observation that the elimination half-life is on the order of 2–3 hours in rats, for example. Renal elimination is generally predicted to be proportional to the fraction of a compound unbound in plasma (e.g., Janků and Zvára (1993)). Transporter-mediated renal resorption would only reduce elimination to a greater extent. If the binding of PFHxA to BSA is indicative of its overall fraction bound in serum and glomerular filtration could remove only 1% (i.e., the free faction) of PFHxA carried in the corresponding serum flow, the elimination half-life should be much longer than is observed. Thus, although plasma protein binding could play some role in PFHxA distribution and elimination, one must be careful in quantitatively interpreting such results. Because it is reversible, protein binding could have a limited impact on distribution and elimination, despite a relatively high fraction of plasma protein binding at equilibrium. Therefore, the empirically determined distribution and elimination rates for PFHxA in various species and sexes are used rather than the rate one might predict based on albumin binding.

3.1.3. Metabolism

Similar to other PFAA compounds, PFHxA is not readily metabolized as evidenced by the findings that no metabolites were recovered from either the liver or urine following oral dosing of mice or rats (Gannon et al., 2011; Chengelis et al., 2009a). Although PFHxA is resistant to metabolism, fluorotelomer-alcohols and sulfonates can undergo biotransformation to form PFHxA or its glucuronide and sulfate conjugates in rodents and humans (Kabadi et al., 2018; Russell et al., 2015).

3.1.4. Elimination

Existing evidence has consistently suggested PFHxA has a shorter half-life than those of other longer chained PFAAs (e.g., PFOA or PFOS). For instance, approximately 80% of the administered dose of PFHxA appeared in the urine of rats 24 hours post-dosing regardless of sex following i.v. injection (Chengelis et al., 2009a). Daikin Industries recovered approximately 90% of an oral dose of 50 mg/kg PFHxA, either as a single dose or on the 14th day of dosing by 24 hours

after the single or last dose in male and female rats and mice (Daikin Industries, 2009a, b). Likewise Dzierlenga et al. (2019) reported that liver and kidney concentrations peaked by 30 min in male rats and by 1 hour in female rats after gavage and decreased steadily thereafter (observations at 0.5, 1, 3, 6, 9 and 12 hours). The tissues concentrations of PFHxA tended to be very low or not quantifiable 24 hours after dosing in both sexes of mice and rats (Iwabuchi et al., 2017; Gannon et al., 2011).

The comparable weight-normalized blood elimination half-life of PFHxA across mammalian species further implies the lack of species-specific roles for renal tissue transporters, either in facilitating elimination or impeding elimination through renal resorption for PFHxA, unlike the situation for some long-chain PFAAs. Gomis et al. (2018) concluded PFHxA had a relatively short elimination half-life and the lowest bioaccumulation among the six PFAAs they evaluated based on applying a one-compartment PK model combined with PK data compiled from previous studies on male rats. In particular, the beta- or elimination-phase half-life ($t_{½,\beta}$) values estimated were: PFHxA = 2.4 hours, perfluorobutane sulfonate (PFBS) = 4.7 hours, pentafluorobenzoic acid (PFBA) = 9.2 hours, ammonium 2,3,3,3-tetrafluoro-2-(heptafluoropropoxy)-propanoate (GenX) = 72 hours, PFOA = 136 hours, and PFOS = 644 hours (Gomis et al., 2018). PK model simulations from a 10-day oral experiment with a dose of 1 mg/kg-day predicted that, as compared to other PFAAs, PFHxA had the lowest serum and liver AUC levels. Likewise, Chengelis et al. (2009a) compared PFHxA dosimetry in naïve male and female rats to results after 25 days of dosing (50–300 mg/kg-day) and found no significant difference in the parameters evaluated, with the serum half-life remaining in the range of 2–3 hours.

Rat Studies

Iwai (2011) evaluated PFHxA excretion in Sprague-Dawley rats and CD-1 mice treated with single and multiple (4 days) oral dose(s) at 50 mg/kg of [^{14}C] ammonium perfluorohexanoate (APFHx). Urine and feces samples were collected for 0–6 hours (urine only) and 6–24 hours and then followed 24-hour intervals until 72 hours after dosing. Expired air was collected over 0–24 and 24–48 hours following oral exposure. For the single dose administration in rats, 97%–100% of administered PFHxA dose was recovered within 24 hours with urine as the major route of elimination (73.0%–90.2%), followed by feces (7.0%–15.5% of the administered dose). No appreciable PFHxA was found in expired air. Two percent of the dose remained in the gastrointestinal tract and carcass. Comparable findings were observed with the multiple oral dose administration (14 daily doses) scenarios (Iwai, 2011).

Chengelis et al. (2009a) reported the terminal half-life of PFHxA in serum was about 2.4-fold shorter for female Sprague-Dawley rats than for male rats (0.42 hours compared to 1.0 hour) with a single dose of 10 mg/kg i.v. injection. Likewise, Gannon et al. (2011) reported elimination half-lives for PFHxA of 1.7 and 1.5 hours in male rats and 0.5 and 0.7 hours in female rats for doses of 2 and 100 mg/kg, respectively. On the other hand, after repeated oral administration (50–300 mg/kg-day) of PFHxA, Chengelis et al. (2009a) found the serum terminal

half-life of PFHxA was generally in the range of 2–3 hours regardless of sex. Comparable urinary elimination half-lives following single 10 mg/kg i.v. were also observed (males: 2.1 hours; females 2.5 hours) (Chengelis et al., 2009a). It is unclear why Chengelis et al. (2009a) obtained different half-lives for males versus females from some of their results, but not in others. Evaluation of the half-life from any PK data set depends on the study design, especially the number and spacing of data points relative to the half-life, the type of PK analysis done, and analytic sensitivity. EPA analyzed PFHxA half-lives that combined data across studies to obtain sex-specific values, described in Section 5.2.1 (Approach for Animal-Human Extrapolation of PFHxA Dosimetry).

As noted above, Daikin Industries evaluated urinary and fecal excretion in Sprague-Dawley rats after 50 mg/kg oral doses for 1 or 14 days (Daikin Industries, 2009a, b). The elimination pattern is consistent with other studies described here, with approximately 90% of the dose recovered in feces and urine by 24 hours. Because excretion was only evaluated at 6 hours (urine only), 24 hours, and multiple days thereafter, these specific studies are not considered quantitatively informative for evaluation of half-life or clearance.

Russell et al. (2015) conducted PK modeling analysis of 3,3,4,4,5,5,6,6,7,7,8,8,8-Tridecafluorooctanol (6:2 FTOH) inhalation (0.5 or 5 ppm) in rats, including its metabolite PFHxA, as described above. The estimated PFHxA half-lives were 1.3 and 0.5 hours in male and female rats, respectively, from single-day exposures, with the estimated yield of PFHxA ranging from 0.5 mol% to 1.9 mol%. The model assumes, however, that the yield of PFHxA from 6:2 FTOH is independent of time. This apparent time-dependence in the half-life could be an artifact of that assumption if induction of metabolism during the dosing period leads to a higher yield with later times. A more comprehensive multiday PK analysis would be needed to demonstrate time-dependent PFHxA clearance unequivocally. Using a noncompartmental PK analysis Kabadi et al. (2018) reanalyzed the 1-day data of Russell et al. (2015) and obtained the same half-life values (1.3 and 0.5 hours in males and females).

A recent study by Dzierlenga et al. (2019) and NTP (2017) showed no apparent pattern in $t_{½,\beta}$ among the i.v. (40 mg/kg) and two lower oral doses (40 and 80 mg/kg) for each sex (ranges 5.74–9.3 hours for male rats and 2.3–7.3 hours for female rats), which likely reflects experimental variability. The $t_{½,\beta}$ for the 160 mg/kg oral dose appeared higher than the other three measurements (13.7 ± 14.2 and 12.2 ± 23.6 hours [mean ± standard error of the mean] for males and females, respectively), but a loss of dose-concordance occurred among the PK data starting at 6 hours (i.e., the serum concentrations were similar for all dose levels at 6 hours and beyond). Also, the data at the last time point (24 hours) varied considerably, resulting in large uncertainty in the estimated terminal half-lives (Dzierlenga et al., 2019).

Similar to the elimination half-life in male Sprague-Dawley rats, the estimated serum elimination half-life of PFHxA in male Wistar rats (6 weeks old) was about 2.6 hours for a single dose of 100 µg/kg BW or 2.9 hours for exposures in drinking water of 1 or 3 months (Iwabuchi et al., 2017). Using a single-compartment PK model with an elimination constant defined as

$k_e = \ln(2)/t_{1/2}$ and obtained from a single-day exposure, the predicted serum concentration after 1 and 3 months of exposure was only 10% higher and 15% lower than the measured concentrations at these time points, respectively. Thus, a systematic change in the half-life or clearance with repeated dosing is not apparent.

In support of the empirical estimates of half-lives described above indicating sex-specific differences in the elimination of PFHxA, the differences can be explained (at least in part) based on available mechanistic information. Specifically, sex hormone-dependent differences occur in expression of transporter proteins in the rat kidney. In rats, kidney Oatp1a1 is expressed at the apical membrane of the proximal tubule (Bergwerk et al., 1996) and mediates sodium-independent transport of thyroid hormones, cholesterol-derived molecules (Hata et al., 2003; Shitara et al., 2002), and PFAS (Han et al., 2012; Yang et al., 2010). In male rats, Oatp1a1 mRNA expression was 2.5-fold greater than in females, undetectable in castrated rats, and inducible in male rats by treatment with estradiol (Kudo et al., 2002).

A separate study (Lu et al., 1996) reported the same sex hormone-dependent effect on Oatp1a1 mRNA expression in castrated males or ovariectomized females treated with testosterone or estradiol. Further, Gotoh et al. (2002) confirmed that Oatp1a1 protein levels were undetectable from female rat kidney and highly expressed in male rat kidney. Because these hormone-dependent transporters are predicted to increase renal resorption of PFHxA in male rats, the implication is that PFAS elimination in female rats should be more rapid compared with male rats. Not all the results above match this expectation, which could reflect a limited activity of the renal transporters toward PFHxA, or simply aspects of experimental design and sampling that measure the PK parameters better in some studies than others. The empirical results of Chengelis et al. (2009a) and Dzierlenga et al. (2019), however, are consistent with this prediction: higher clearance and shorter half-lives in female rats compared to male rats.

Some evidence also suggests the affinity for Oatp1a1 depends on PFAS chain length. Specifically, Yang et al. (2009) examined the role of PFAS (C4–C12) in inhibiting the uptake of estrone-3-sulfate (E3S3S) using Oatp1a1-expressing Chinese hamster ovary cells. They showed the level of inhibition of E3S uptake increased as the chain length increased; for example, PFHxA inhibited E3SS uptake with an inhibition constant (K_i) of 1,858 µM, as compared with 84 µM for PFOA. This high K_i for PFHxA (i.e., the concentration required to inhibit one-half the Oatp1a1 activity, 584 µg/mL) indicates a low affinity of PFHxA for the transporter and thus leads to predictions of a low impact of Oatp1a1 expression on PFHxA elimination kinetics, contrary to the empirical PK data discussed above. Chengelis et al. (2009a) clearly showed more rapid elimination in female rats versus male rats at serum concentrations below 40 µg/mL, that is, an order of magnitude or more below the K_i. As most of the water is resorbed from the renal filtrate, however, the concentration of PFHxA in the remaining fluid will increase proportionately. Thus, the PFHxA concentrations in the proximal tubule of these rats (where Oatp1a1 is expressed) could be high enough for significant transporter activity, but below the level of saturation.

Collectively, the evidence provides a biologically plausible explanation for the observed sex-specific PFHxA elimination in rats (i.e., the 2- to 3-fold longer half-life in male versus female rats), although uncertainties remain (Han et al., 2012; Gannon et al., 2011; Chengelis et al., 2009b). Most notably, whether this apparent sex difference in re-uptake exists in humans or in species other than rats is unclear. Organic-anion transporters are known to be under hormonal regulation in rat and mouse kidney, with gender-specific differences in their expression likely regulated by sex-hormone receptors. Some evidence suggests similar sex-related differences in humans (Sabolić et al., 2007). Kudo et al. (2001) demonstrated that the sex-related difference in PFOA elimination in rats was abolished when male rats were castrated, increasing to match that in females, and that its elimination was reduced in both females and castrated males treated with testosterone. This demonstration of hormone-related elimination for PFOA and observations of sex differences in the elimination of other PFAS such as PFNA, PFOA, and PFBS (Chengelis et al., 2009a; Kudo et al., 2001) suggest this is a common underlying mechanism for PFAS elimination.

Mouse Studies

As stated above (Elimination, Rat Studies), Iwai (2011) evaluated PFHxA excretion in CD-1 mice after single and 14-day oral exposures. Results were similar for single and multiple dose administrations. After multiple doses, >95% of the administered PFHxA was recovered within 24 hours with urine as the major route of elimination (77.8%–83.4%), followed by feces (9.6%–12.9% of the administered dose). Only 0.6%−0.9% remained in the gastrointestinal tract and carcass. Gannon et al. (2011) also evaluated PFHxA PK in mice but state they did not report half-lives in mice because the data showed a biphasic clearance pattern that precluded use of the standard noncompartmental modeling.

As noted above, Daikin Industries evaluated urinary and fecal excretion in CD-1 mice after 50 mg/kg oral doses for 1 or 14 days (Daikin Industries, 2009a, b). The elimination pattern is consistent with Iwai (2011), with approximately 90% of the dose recovered in the urine and feces (total) after 24 hours. Because excretion was only evaluated at 6 hours (urine only), 24 hours, and multiple days after the PFHxA dosing ended, however, the studies cited are not considered quantitatively informative for evaluation of half-life or clearance.

Daikin Industries (2010) evaluated the time-course of PFHxA in female Crl:CD(1CR) mouse plasma after single oral gavage doses of 35, 175, and 350 mg/kg, with concentrations measured at 0.5, 2, 4, 6, 8, and 24 hours. The estimated half-life was between 0.9 and 1.2 hours for the three dose groups but lacked a dose-dependent pattern. However, the C_{max}/dose was 2.76, 1.88, and 1.30 kg/L for the 35, 175, and 350 mg/kg doses, respectively, indicating saturation of absorption with higher doses. The $AUC_{0-\infty}$/dose was not dose-dependent, although it varied between 5.1 and 6.5 kg-h/L, indicating that clearance was not dose-dependent.

The plasma time-course data from Gannon et al. (2011) and Daikin Industries (2010) were reevaluated by EPA as described with the derivation of the HED in Section 5.2.1 (Approach for

Animal-Human Extrapolation of PFHxA Dosimetry) and Appendix C to obtain overall pharmacokinetic parameters.

Monkey Studies

In the study on cynomolgus monkeys by Chengelis et al. (2009a), three males and three females received 10 mg/kg PFHxA by i.v. injection. The mean clearance was nearly the same in both sexes (0.122 L/h-kg in males and 0.136 L/h-kg in females), but the estimated half-life appeared longer in males (5.3 ± 2.5 hours) than in females (2.4 ± 1.7 hours) with a corresponding apparent difference in V_d (0.989 L/kg in males and 0.474 L/kg in females). The similarity of the clearance values and the nearly identical serum values for males and females after the first 4 hours suggest no striking sex differences in the pharmacokinetics of PFHxA in monkeys.

Human Studies

No controlled exposure PK studies of PFHxA elimination in humans are available but Russell et al. (2013) applied PK analysis to biomonitoring data from Nilsson et al. (2013) to estimate the half-life of PFHxA in humans. Specifically, Russell et al. (2013) estimated the apparent half-life of PFHxA in humans by analyzing biomonitoring data collected from professional ski wax technicians and then compared the human estimates of PFHxA elimination to that for mice, rats, and monkeys. For the human monitoring study, blood samples (n = 94) were collected from male professional ski wax technicians (n = 11) and analyzed for PFHxA in plasma and serum. (Individual data for eight of the technicians are shown in Appendix C.2; complete data are available as the supplemental information for Nilsson et al. (2013)). Personal and area air concentration monitoring of the ski wax subjects and facilities demonstrated both the metabolic precursor, 6:2 FTOH, and PFHxA were present in all locations, but the arithmetic mean concentration of 6:2 FTOH ranged from over 100 times higher than PFHxA to almost 100 times lower, across the monitoring locations. A one-compartment model with first-order kinetics was used for PK analyses. The estimated geometric mean half-life of PFHxA was 32 days with a range of 14–49 days in the studied population (Russell et al., 2013). PFHxA plasma concentrations declined below the plasma detection limit of 0.05 ng/mL within a period of 2–4 months after exposure ceased, reflecting the relatively rapid elimination rate of PFHxA. In contrast, the half-life of PFHxS in humans was estimated to range from 5 to 9 years (Olsen et al., 2007).

Analysis by Luz et al. (2019) found no significant species- or sex-related differences in the elimination kinetics of PFHxA. The PK analysis, however, is attributed to a meeting abstract (Buck and Gannon, 2017) and provides no details of the methods the authors used. The text of Luz et al. (2019) indicates the analysis of Buck and Gannon (2017) used data from only 3 of the 11 subjects of Nilsson et al. (2013), specifically the 3 with the most rapid elimination, reducing the extent to which the conclusion can be reliably extrapolated to the population as a whole. Luz et al. (2019) state slower apparent elimination could occur in some subjects because of ongoing exposure. Although ongoing exposure could cause this effect, it is also possible that elimination in some individuals is

slower than others due to interindividual variability. In the absence of independent evidence that ongoing exposure occurred in other human subjects of Nilsson et al. (2013) who were excluded in this later analysis, EPA does not consider basing conclusions on human elimination on only the three individuals who had the most rapid elimination appropriate.

EPA examined the data of Nilsson et al. (2013), and the observed seasonal variation appears to show a longer systemic period of exposure (when blood levels are elevated versus. Declining) for some individuals than others. Also, the data set includes samples with concentrations below the limit of detection (LOD) that should be treated with an appropriate statistical model to account for the censoring of these data. Finally, only the data collected encompassing the 2007–2008 ski season, during which only 8 of the 11 technicians were sampled, includes post-exposure samples needed to quantify elimination. A detailed description of EPA's analysis of the eight technicians sampled during the 2007–2008 season is provided in Appendix C.2. Briefly, each ski-wax technician in the study was presumed to have a constant rate of exposure up to a date that is different for each individual when exposure stopped, and elimination began. Specifically, we used a one-compartment i.v.-infusion model to fit the data:

$$C(t) = \begin{cases} \frac{A}{tinf \cdot ke} \cdot \left(1 - e^{-ke \cdot t}\right), & if\ t \le tinf \\ \frac{A}{tinf \cdot ke} \cdot \left(1 - e^{-ke \cdot tinf}\right) \cdot e^{-ke \cdot (t - tinf)}, & if\ t > tinf \end{cases} \quad (3\text{-}2)$$

Where $A = dose/V_d$, *tinf* is the time period of exposure (treated as an infusion), and *ke* is the elimination rate. The model is analyzed through hierarchical Bayesian analysis, with *A* and *tinf* estimated independently for each individual technician although the technician-level *ke* is drawn from a population-level distribution. Note blood concentrations were measured only once a month and no other data on exposure is available. Thus, although the model clearly simplifies the exposure estimation, estimating a larger number of parameters reliably would not be possible. As such, the model allows for estimating variation among individuals without subjectively selecting a subset of the technicians for analysis. The resulting distribution of *ke* had a mean (90% confidence interval, CI) of 0.00252 (0.00136–0.00477) h^{-1}. Using an average V_d of 0.7315 L/kg (731.5 mL/kg) for male and female monkeys from Chengelis et al. (2009a), the resulting mean for human clearance is $CL = V_d \cdot ke$ = 1.84 mL/kg-h. Given the expected similarity of V_d across mammalian species, EPA considers the average value estimated for rats (0.33 L/kg) to be a reasonable lower bound for humans and the highest value reported by Chengelis et al. (2009a) for an individual (male) monkey (1.54 L/kg) to be a reasonable upper bound. Combining these with the 90% CI for *ke* (0.00136–0.00477 h^{-1}) yields a possible range for human clearance of 0.45–7.35 mL/kg-h, a range of 16-fold from 4-fold above to 4-fold below the estimated mean.

Xiao et al. (2011) measured the serum concentrations of 10 PFAA chemicals in 227 nonoccupationally exposed individuals aged 0.3–90 years (133 males and 94 females) in China. Significant positive correlations were observed between age and serum levels of PFAA chemicals

except for PFBS, PFHpA, and PFHxA. Spearman correlation coefficients between age and serum PFHxA were 0.20, −0.02, and 0.08 for males, females, and the combined data, respectively. Collectively, the findings indicated no age-related accumulation of PFHxA in human bodies, which is consistent with the relatively short half-life.

3.1.5. PBPK Models

No PBPK model is available for PFHxA in rats, mice, or monkeys. Fàbrega et al. (2015) described a PBPK model for multiple PFAS in humans, including PFHxA. However, Fàbrega et al. (2015) state two key parameters that determine the rate of resorption from glomerular filtrate in the kidney were identified using the data from the Ericson et al. (2007) epidemiological survey of PFAS exposure in residents of Catalonia, Spain. Because PFHxA was not detected in any individuals sampled by Ericson et al. (2007), EPA does not consider it possible to reliably identify elimination parameters from that data set. Further, the individual exposure or elimination data needed to associate the blood concentrations of Ericson et al. (2007) with urinary clearance rates are *not reported* in either paper. Thus, uniquely identifying two parameters with a single combination of average PFHxA exposure and average blood concentration is impossible. Finally, as described above (Distribution, Distribution in Humans), the tissue: blood partition coefficients Fàbrega et al. (2015) estimated are not considered suitable for the purposes of this assessment due to the 4+-year lag in measurements between collection of the blood samples and the tissue samples and because they are inconsistent with data on PFHxA distribution in other species, including monkeys. Thus, the PBPK model of Fàbrega et al. (2015) is not considered sufficiently suitable for use in this assessment.

3.1.6. Summary

The PFHxA elimination half-lives and clearance values reported in studies are important for interpreting and quantifying health outcomes potentially associated with PFHxA exposure. The most notable finding was the apparent sex-specific PK differences between male and female mice and rats where female rodents eliminate PFHxA 2–3 times faster than males (see Table 3-1). Although monkeys have half-lives and clearance values in the same range as mice and rats, the clearance in female monkeys is only 11% faster than in males. This indicates that the significant sex differences observed in rodents does not appear to apply to primates. Humans have a much longer serum elimination half-life (EPA estimate: 275 hours) than rodents and monkeys (2–7 hours). The difference could be a consequence of species differences in the expression or activity of the renal transporters that reabsorb PFAS, but this has not been demonstrated. All available PK evidence is summarized below in Table 3-1.

According to EPA's $BW^{0.75}$ guidelines (U.S. EPA, 2011), use of chemical-specific data for dosimetric extrapolation such as the PFHxA-specific data described above is preferable to the default method of $BW^{0.75}$ scaling. That is the case here. For example, using the standard species BWs of 0.25 kg in rats and 80 kg in humans, the half-life in humans is predicted to be 4.2 times greater

than rats. Given half-lives in the range of 0.4–14 hours among male and female rats (see Table 3-1), one would then predict half-lives of 1.6–57 hours in humans, 20–200 times lower than the range estimated by Russell et al. (2013) and 10–100 times lower than the range estimated by EPA (see Table 3-1). Thus, based on the PFHxA-specific PK data, use of $BW^{0.75}$ for dosimetric extrapolation could lead to an underprediction of human elimination by 1–2 orders of magnitude. Therefore, use of $BW^{0.75}$ as an alternative means of extrapolation is not considered further for PFHxA, and the preferred, data-driven approach will be used for the dosimetric extrapolation.

Table 3-1. Summary of PK evidence for PFHxA

Species/sex	Study design (dose)	Elimination half-life (beta) (h)	AUC/dose (kg-h/L)	Clearance (mL/h-kg)	Volume of distribution (mL/kg)	Reference
Rats						
Male	Single i.v. dose (10 mg/kg)	1.0	8.7	116	175	Chengelis et al. (2009a)
	Single oral dose (50 mg/kg)	2.2	10.0	NR	NR	
	Single oral dose (150 mg/kg)	2.4	6.1	NR	NR	
	Single oral dose (300 mg/kg)	2.5	8.4	NR	NR	
Female	Single i.v. dose (10 mg/kg)	0.42	1.3	775	466	
	Single oral dose (50 mg/kg)	2.6	2.4	NR	NR	
	Single oral dose (150 mg/kg)	2.2	2.2	NR	NR	
	Single oral dose (300 mg/kg)	2.1	3.5	NR	NR	
Male	Single i.v. dose (40 mg/kg)	8.0 ± 2.2	7.4 ± 0.7	136 ± 13	430 ± 112	Dzierlenga et al. (2019) NTP (2017)
	Single oral dose (40 mg/kg)	9.3 ± 20.8	9.7 ± 1.3	103 ± 13	601 ± 470	
	Single oral dose (80 mg/kg)	5.7 ± 4.6	6.6 ± 0.5	153 ± 11	496 ± 81	
	Single oral dose (160 mg/kg)	14 ± 14	6.8 ± 0.6	147 ± 14	615 ± 367	
Female	Single i.v. dose (40 mg/kg)	7.3 ± 2.0	3.1 ± 0.3	327 ± 33	223 ± 45	
	Single oral dose (40 mg/kg)	2.3 ± 213	6.1 ± 1.1	164 ± 29	327± 149	
	Single oral dose (80 mg/kg)	5.5 ± 2.6	3.2 ± 0.4	314 ± 39	560 ± 113	
	Single oral dose (160 mg/kg)	12 ± 24	3.7 ± 0.5	274 ± 37	473 ± 158	
Male	Single oral dose (2 mg/kg)	1.7 ± 0.6	8 ± 1.5	NR	NR	Gannon et al. (2011)
	Single oral dose (100 mg/kg)	1.5 ± 0.2	6.5 ± 1.4	NR	NR	
Female	Single oral dose (2 mg/kg)	0.5 ± 0.1	2.5 ± 0.5	NR	NR	
	Single oral dose (100 mg/kg)	0.7 ± 0.3	2.5 ± 0.7	NR	NR	

Species/sex	Study design (dose)	Elimination half-life (beta) (h)	AUC/dose (kg-h/L)	Clearance (mL/h-kg)	Volume of distribution (mL/kg)	Reference
Male	Single i.v. dose (0.1 mg/kg)	2.7	9.8	NR	400	Iwabuchi et al. (2017)
Male	Single inhalation[a] (0.5 ppm)	1.3	ND[b]	107	NR	Kabadi et al. (2018)
	Single inhalation[a] (5.0 ppm)	1.3	ND[b]	277	NR	
Female	Single inhalation[a] (0.5 ppm)	0.5	ND[b]	107	NR	
	Single inhalation[a] (5.0 ppm)	0.5	ND[b]	277	NR	
Mice						
Male	Single oral dose (2 mg/kg)	ND	12	NR	NR	Gannon et al. (2011)
	Single oral dose (100 mg/kg)	ND	12	NR	NR	
Female	Single oral dose (2 mg/kg)	ND	4	NR	NR	
	Single oral dose (100 mg/kg)	ND	6.4	NR	NR	
Monkeys						
Male	Single i.v. dose (10 mg/kg)	5.3 ± 2.5	8.4 ± 1.8	122 ± 24	989 ± 579	Chengelis et al. (2009a)
Female	Single i.v. dose (10 mg/kg)	2.4 ± 1.7	7.5 ± 1.3	136 ± 22	474 ± 349	
Humans						
Males and females	Post-exposure observation in male ski wax technicians	768 (336–1,176) 275 (145–509)	ND	ND 1.84 (0.45–7.35)	ND	Russell et al. (2013) Current analysis

i.v. = intravenous; ND = not determined; NR = not reported.

[a]6-hour inhalation exposure to 6:2 FTOH.

[b]Dose of PFHxA unknown.

3.2. NONCANCER EVIDENCE SYNTHESIS AND INTEGRATION

For each potential health effect discussed below, the synthesis describes the evidence base of available human and animal studies. The PFHxA animal literature inventory summarizes the evidence base on potential health effects (organized by organ or system) from the available *high* and *medium* confidence short-term, developmental, subchronic, and chronic studies in mice and rats (NTP, 2018; Klaunig et al., 2015; Iwai and Hoberman, 2014; Chengelis et al., 2009b; Loveless et al., 2009; Kirkpatrick, 2005a). These animal studies represent the primary evidence available for this PFAS, and a more detailed summarization of study methods and findings is provided in HAWC. Some organs/systems for which animal data were available are summarized in the animal literature inventory, but these data were not synthesized due to insufficient evidence to draw hazard judgments (i.e., **evidence** is **inadequate**). Specifically, for these health effects there were either few studies with null results (i.e., dermal, musculoskeletal, sensory, ocular) or few studies with sporadic findings of unclear toxicological significance (i.e., respiratory, gastrointestinal system, cardiovascular, and metabolic effects), including small changes in indirect outcome measures and other effects of unclear biological significance in isolation (e.g., decreases in cholesterol). Similarly, one *low* confidence cross-sectional study in general population adults in China examined the association between PFHxA exposure and adiposity (Tian et al., 2019); however, due to concern for exposure misclassification resulting from reverse causation, no judgment could be drawn (i.e., **evidence** is **inadequate**) and these data were not synthesized. Effects on body weights and survival, which had no effect or lowest effect levels at the highest administered dose in animal studies, were also not separately synthesized but were used to aid the interpretation of other potential health effects of PFHxA exposure.

3.2.1. Hepatic Effects

Human

Three epidemiological studies report on the relationship between PFHxA exposure and liver enzymes. Of these, one (Jiang et al., 2014), a cross-sectional study of pregnant women in China, was critically deficient in the confounding domain and was considered overall *uninformative*. There was no consideration of potential confounding in the study design and analysis. Most notably, there was no adjustment for age, which is a highly relevant potential confounder of the association. Based on these deficiencies, the study was excluded from further analysis (see Figure 3-1). The remaining studies (Liu et al., 2022; Nian et al., 2019) were cross-sectional studies in general population adults in China and were classified as *medium confidence* (see Figure 3-1). Liu et al. (2022) reported positive associations (i.e., higher liver enzyme levels with higher PFHxA exposure) for serum albumin ($p < 0.05$) and alanine aminotransferase (ALT) and alkaline phosphatase (ALP) (not statistically significant), but no association with aspartate aminotransferase (AST), total protein, or γ-glutamyl transferase (GGT). Nian et al. (2019) did not observe an association between PFHxA

levels and ALT, AST, total protein, ALP, GGT, total bilirubin, or cholinesterase. Sensitivity was a concern in both studies due to limited exposure contrast (i.e., likely insufficient variability in exposure levels to detect an association), which may explain the limited observed associations.

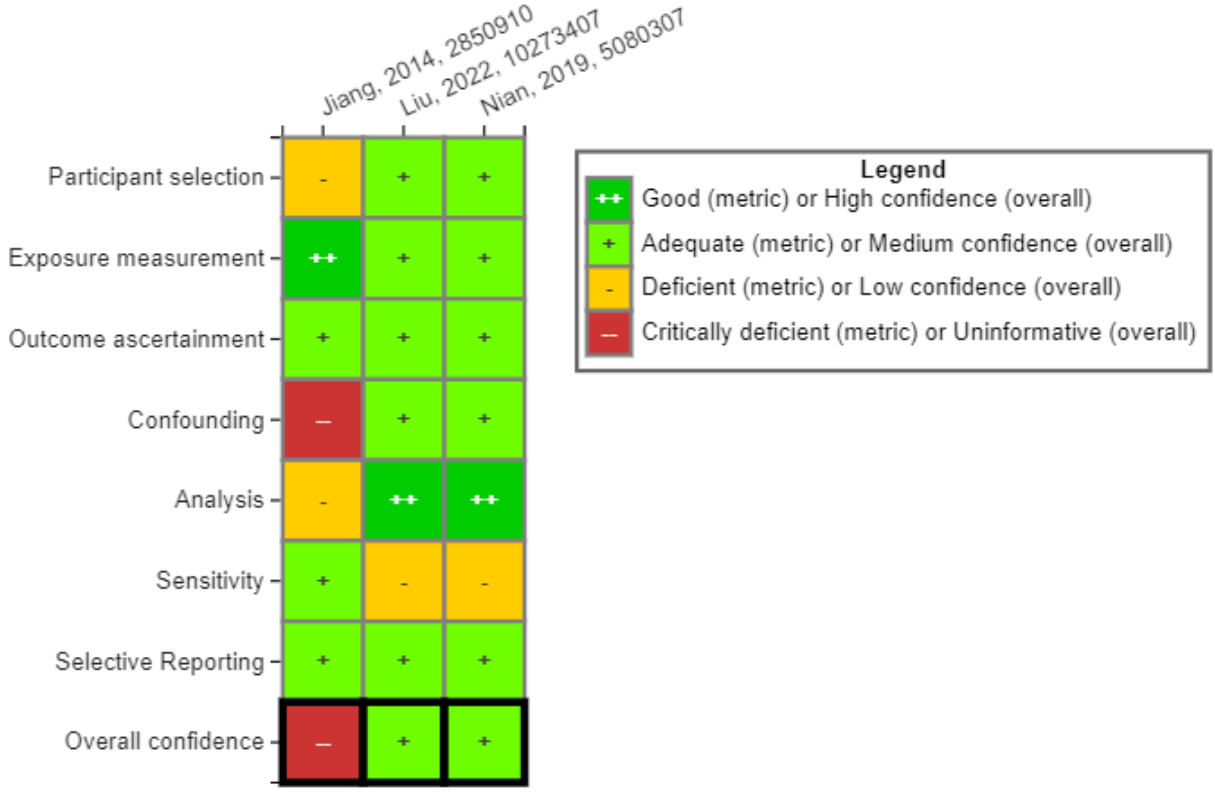


Figure 3-1. Study evaluation for human epidemiological studies reporting hepatic system findings from PFHxA exposures (full details available by clicking the HAWC link). Note that for N/A, critical deficiencies in confounding domains were identified and the study was judged as *uninformative*; thus, the remaining domains were not evaluated.

Animal

Hepatic outcomes were evaluated in multiple short-term, subchronic, or chronic studies in rats and mice (NTP, 2018; Klaunig et al., 2015; Iwai and Hoberman, 2014; Chengelis et al., 2009b; Loveless et al., 2009). Generally, studies were rated as *medium or high* confidence for the hepatic outcomes, but some outcome-specific considerations for study evaluation were influential on the overall confidence ratings for hepatic effects. Histopathology for Chengelis et al. (2009b) was rated *low* confidence because of issues related to observational bias, endpoint sensitivity and specificity, and results presentation. Results of the outcome-specific confidence evaluations are presented in Table 3-2 below, and details are available by clicking the HAWC link.

Table 3-2. Evaluation results for animal studies assessing effects of PFHxA exposure on the hepatic system

Author (year)	Species, strain (sex)	Exposure design	Exposure route and dose range	Organ weight	Histopathology	Clinical chemistry	Peroxisomal beta oxidation
NTP (2018)	Rat, Harlan Sprague-Dawley (male and female)	Short term (28 d)	Gavage[a] Male and female: 0, 62.5, 125, 250, 500, 1,000 mg/kg-d	++	++	++	NM
Chengelis et al. (2009b)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	Subchronic (90 d)	Gavage[a] Male and female: 0, 10, 50, 200 mg/kg-d	++	–	++	–
Loveless et al. (2009)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	Subchronic (90 d)	Gavage[b] Male and female: 0, 20, 100, 500 mg/kg-d	++	++	++	++
Klaunig et al. (2015)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	2-yr cancer bioassay	Gavage[a] Male: 0, 2.5, 15, 100 mg/kg-d Female: 0, 5, 30, 200 mg/kg-d	NM	++	++	NM

++ Outcome rating of *high* confidence; + outcome rating of *medium* confidence; - outcome rating of *low* confidence; – outcome rating of *uninformative*; NM, outcome not measured.

[a,b]Study evaluation for animal toxicological hepatic endpoints reported from studies with male and female rats receiving by gavage PFHxA[a] or PFHxA sodium salt. [b] Study evaluation details for all outcomes are available by clicking the HAWC link.

Organ Weight

Overall, findings of increased liver weights after oral PFHxA or PFHxA sodium salt exposures in rats were consistent (see Figure 3-2; exposure response array link). Relative liver weights (see Table 3-3) are generally considered more reliable than absolute liver weights because they consider large variations in body weight that could skew organ weight interpretation (Hall et al., 2012). Large variations in body weights were not observed after PFHxA exposures in male and female adult rats, and changes in both relative and absolute liver weights were similarly increased and dose responsive. Increases in relative and absolute liver weights were dose-dependently increased in all three short-term and subchronic studies. Statistically significant increases in male rat relative liver weights were observed with oral doses of ≥200–250 mg/kg-day, whereas statistically significant increases in female rats were observed only at ≥500 mg/kg-day. Specifically, in the 28-day study, relative liver weight was increased (14%) in male rats at 250 mg/kg-day, where a similar increase (15%) was observed in female rats at 500 mg/kg-day (NTP, 2018). In the subchronic studies, relative liver weights were increased (22%) at 200 mg/kg-day in males (with

no change in females) in one study (Chengelis et al., 2009b), and the other study observed increases of 63% and 37% at 500 mg/kg-day in males and females, respectively (Loveless et al., 2009). Note that PFHxA effects on relative liver weights had resolved by 30 days in the recovery group (Chengelis et al., 2009b). Liver weights were not evaluated in the chronic study (Klaunig et al., 2015).

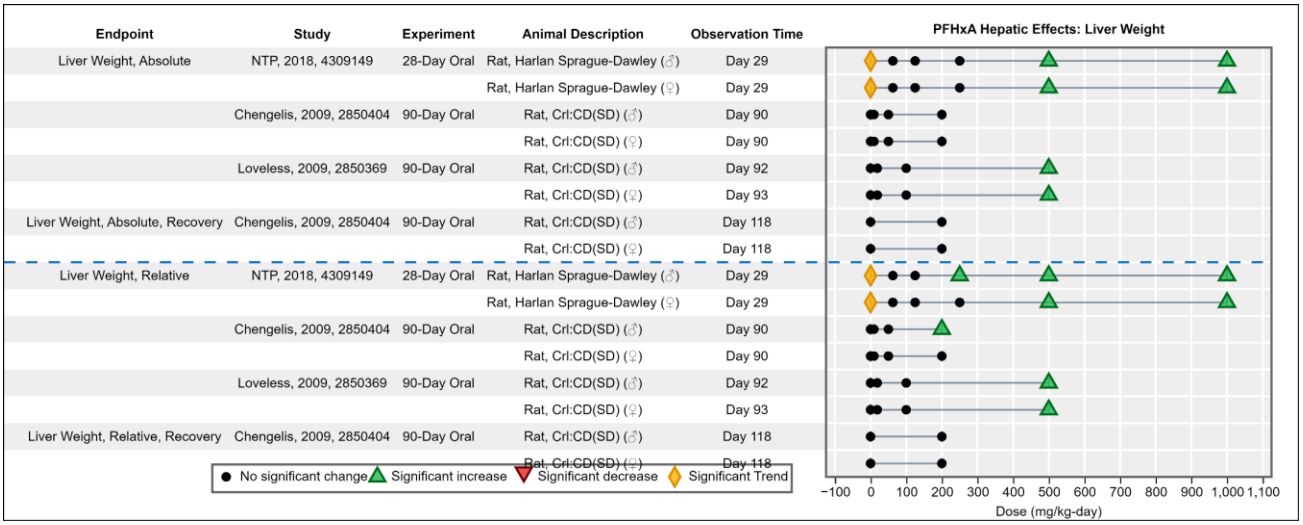


Figure 3-2. Liver weights (absolute and relative) after short-term and subchronic PFHxA exposures (full details available by clicking the HAWC link).

Table 3-3. Percent increase in relative liver weight due to PFHxA exposure in short-term and subchronic oral toxicity studies

Study design and reference	Dose (mg/kg-d)															
	2.5	5	10	15	20	30	35	50	62.5	100	125	175	200	250	500	1,000
28-d female rat (NTP, 2018)									1		2			7	15*	47*
28-d male rat (NTP, 2018)									8		7			14*	32*	64*
90-d female rat (Chengelis et al., 2009b)			4					6					5			
90-d male rat (Chengelis et al., 2009b)			1					1					22*			
90-d female rat (Loveless et al., 2009)					−1					5					37*	
90-d male rat (Loveless et al., 2009)					0					11					63*	

* Indicates instances where statistical significance ($p < 0.05$) compared to controls was reported by study authors; shaded cells represent doses not included in the individual studies.

Histopathology

Treatment-related increases in liver weight can result from various changes in hepatic morphology including hyperplasia of any resident liver cell type, cellular hypertrophy, inflammation, fibrosis, increase in hepatocyte size, neoplasia, congestion, or metabolic enzyme induction (Hall et al., 2012; Thoolen et al., 2010; U.S. EPA, 2002a). As shown in Table 3-4 and summarized in the HAWC link, four studies evaluated liver histopathology in rats. One observed effect of PFHxA exposure was hepatocellular hypertrophy that was consistent across the short term and subchronic studies. Hepatic hypertrophy can refer to an increase in liver weight and size; an increase in hepatocyte size caused by abnormal storage of water, glycogen, lipids, or organelle proliferation; and an increase in hepatic enzyme induction (Hall et al., 2012; Thoolen et al., 2010; U.S. EPA, 2002a). Coherent with findings on liver weight, the observations of hepatocellular hypertrophy were dose-dependent and male rats were more sensitive than females. Specifically, increased hepatocellular hypertrophy was observed in adult male and female rats in the *high* confidence short-term (NTP, 2018) and *high* confidence subchronic (Loveless et al., 2009) studies at doses ≥100–500 mg/kg-day. In the subchronic study, hypertrophy persisted 30 and 90 days after recovery in males, and 30 days after recovery in females (Loveless et al., 2009). In the *low* confidence (for histopathology outcomes) subchronic study, centrilobular hepatocellular hypertrophy was observed in male rats only (incidence 7/10, 200 mg/kg-day) and resolved after 28-day recovery (Chengelis et al., 2009b). In the chronic study (Klaunig et al., 2015), hepatocellular hypertrophy findings were null consistent with null observations at similar doses in the short-term and subchronic studies.

Table 3-4. Incidence of hepatocellular hypertrophy findings in adult rats due to PFHxA exposure in short-term and subchronic oral toxicity studies

Study design and reference	Dose (mg/kg-d)									
	10	20	50	62.5	100	125	200	250	500	1,000
28-d, female rat (NTP, 2018)				0/10		0/10		0/10	0/10	9/10*
28-d, male rat (NTP, 2018)				0/10		0/10		0/10	9/10*	10/10*
90-d, female rat (Chengelis et al., 2009b)	0/10		0/10				0/10			
90-d, male rat (Chengelis et al., 2009b)	0/10		0/10				7/10*			
90-d, female rat (Loveless et al., 2009)		0/10			0/10				5/10*	
90-d, male rat (Loveless et al., 2009)		0/10			4/10*				10/10*	
90-d, female rat, 30-d recovery (Loveless et al., 2009)									4/10*	

Study design and reference	Dose (mg/kg-d)									
	10	20	50	62.5	100	125	200	250	500	1,000
90-d, female rat, 90-d recovery (Loveless et al., 2009)									0/10	
90-d, male rat, 30-d recovery (Loveless et al., 2009)									9/10*	
90-d, male rat, 90-d recovery (Loveless et al., 2009)									6/10*	

* Indicates instances where statistical significance ($p < 0.05$) compared to controls was reported by study authors; shaded cells represent doses not included in the individual studies.

Other pathological findings of PFHxA-mediated hepatic effects included increased hepatocellular necrosis in rats, with a significant increase in female rats ($n = 99/10$ vs. 0/10 in controls, $p < 0.05$) reported in a short-term study at 1,000 mg/kg-day PFHxA, whereas there were no significant findings of hepatocellular necrosis across male dose groups (NTP, 2018). One subchronic study reported necrosis in male rats ($n = 1/10$ vs. 0/10 in controls, not statistically significant) (Chengelis et al., 2009b) whereas necrosis was not observed in the other subchronic study (Loveless et al., 2009). In the *high* confidence chronic study, Klaunig et al. (2015) reported hepatocellular necrosis ($n = 12/70$ vs. 2/60 in controls, $p < 0.05$ in the 200 mg/kg-day female dose group (the highest dose tested). The authors noted most necrosis findings were in animals that died or were euthanized prior to scheduled necropsy and the increased mortality was not treatment related, but was due to mechanical injury, gavage trauma, reflux injury, or spontaneous disease processes (Klaunig et al., 2015). The authors reported no treatment-related increases in hepatocellular necrosis ($n = 6/70$ vs. 4/60 in controls) or necrosis in the centrilobular regions of the liver lobule ($n = 1/70$ vs. 2/60 in controls) in male rats up to the highest dose for that sex, 100 mg/kg-day. Other findings included nonsignificant congestion in males ($n = 23/70$ vs. 15/60 in controls) and females ($n = 8/70$ vs. 6/60 in controls) (Klaunig et al., 2015). Incidence of necrosis were not observed in the short-term study (NTP, 2018), and the subchronic study by Loveless et al. (2009) did not report histological findings other than hepatocellular hypertrophy (no data on necrosis were available).

Other histopathological findings included observations of hepatocellular cytoplasmic alterations ($p < 0.05$) in adult male and female rats at the highest dose [1,000 mg/kg-day in the short-term study (NTP, 2018)]. All results reported above can be viewed using the HAWC link.

Clinical Chemistry

A clinical chemistry panel measures the proteins, enzymes, chemicals, and waste products in the blood. These measures, when evaluated together and with other biomarkers are informative

diagnostic tests of organ function and when interpreted together with histopathology are useful for the assessment of adverse liver effects.

Serum Enzymes

Alanine aminotransferase (ALT) and aspartate aminotransferase (AST) are often useful indicators of hepatic enzyme induction or hepatocellular damage as increased serum levels are thought to be due to hepatocyte damage resulting in release into the blood, whereas ALP is localized to the bile canalicular membrane and more indicative of hepatobiliary damage (Hall et al., 2012; Amacher et al., 1998). PFHxA effects on the serum enzymes ALT, AST, and ALP included <2-fold increases in serum enzyme across the three short-term and subchronic studies, except for one 2.4-fold increase in male rats at 200 mg/kg-day in the *high* confidence subchronic study (Chengelis et al., 2009b). No clear pattern of effects on the serum enzymes were reported in the chronic study (Klaunig et al., 2015), but the highest dose was 100 or 200 mg/kg-day PFHxA in male or female rats, respectively. Full study details are available in Figure 3-3 and by clicking the HAWC link. Percent changes in treated relative to controls are provided in Table 3-5, Table 3-6, and Table 3-7.

Specifically, in the short-term study, ALT, AST, and ALP were increased in a dose-response gradient in adult male rats at doses as low as 500 mg/kg-day (NTP, 2018). In female rats, ALT and AST measures were increased in a dose-response gradient at doses as low as 500 mg/kg-day, whereas ALP was increased only in the highest (1,000 mg/kg-day) dose group (NTP, 2018).

ALT increases were observed only in male rats at PFHxA sodium salt exposures as low as 20 mg/kg-day in one subchronic study (Loveless et al., 2009) and in the highest PFHxA dose group (200 mg/kg-day) in the other subchronic study (Chengelis et al., 2009b). AST was increased in only one subchronic study in males at ≥20 mg/kg-day (Loveless et al., 2009). Chengelis et al. (2009b) reported increased AST in males only in the 200 mg/kg-day dose group that resolved after the 30-day recovery (see Table 3-6).

ALP was increased in both subchronic studies with significant increases observed in the highest exposure groups [200 (Loveless et al., 2009) and 500 mg/kg-day (Chengelis et al., 2009b)] that resolved by the 30-day recovery (see Table 3-7). The chronic study did not include a 13-week endpoint that would have been useful for group mean comparisons with the test measures in the subchronic studies (as clinical pathology test values often change with animal age) (AACC, 1992).

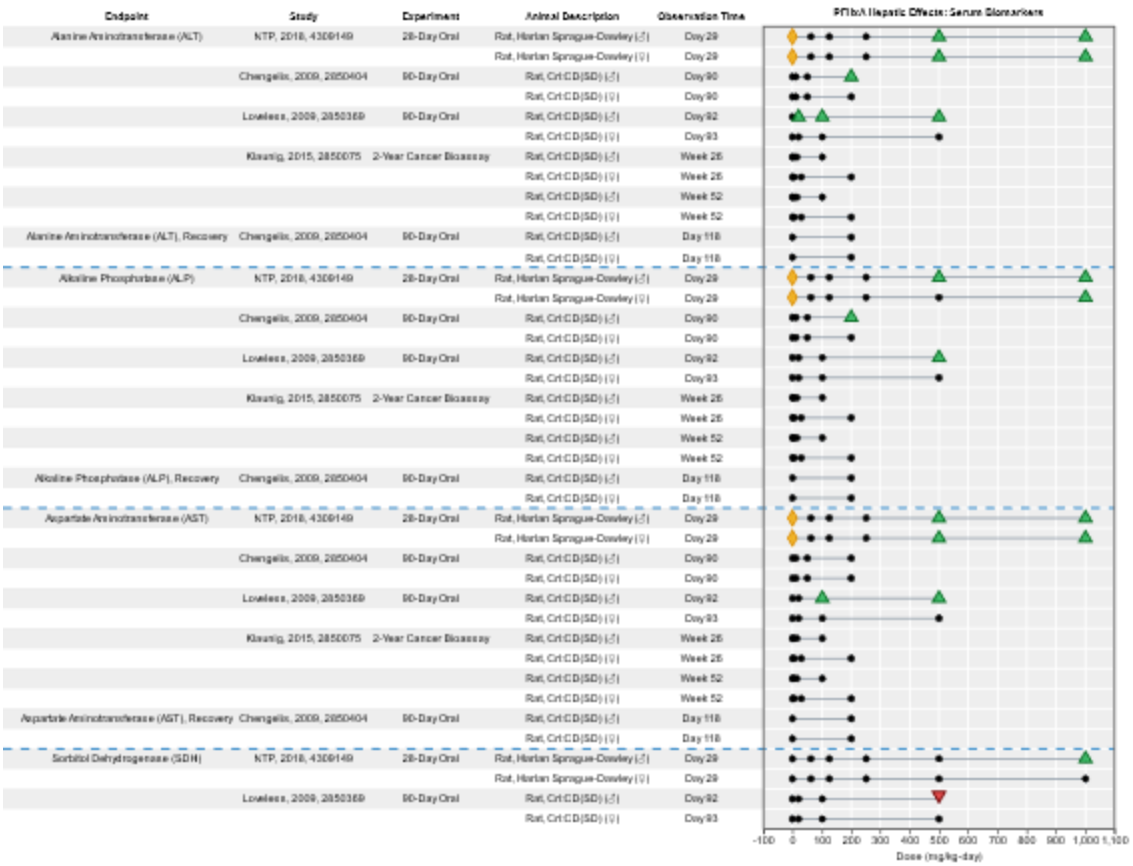


Figure 3-3. Clinical chemistry findings (serum enzymes) after short-term, subchronic, and chronic PFHxA exposures (full details available by clicking the HAWC link).

Table 3-5. Percent change in alanine aminotransferase due to PFHxA exposure in short-term, subchronic, and chronic oral toxicity studies

Study design and reference	Dose (mg/kg-d)													
	2.5	5	10	15	20	30	50	62.5	100	125	200	250	500	1,000
28-d, female rat (NTP, 2018)								11		15		10	35*	44*
28-d, male rat (NTP, 2018)								4		4		8	26*	64*
90-d, female rat (Chengelis et al., 2009b)			60				29				3			
90-d, male rat (Chengelis et al., 2009b)			12				22				237*			
90-d, female rat (Loveless et al., 2009)					−46				−25				−4	

Study design and reference	Dose (mg/kg-d)													
	2.5	5	10	15	20	30	50	62.5	100	125	200	250	500	1,000
90-d, male rat (Loveless et al., 2009)					33*				44*				56*	
Wk 26, female rat (Klaunig et al., 2015)		44				−62					−57			
Wk 26, male rat (Klaunig et al., 2015)	10			12					117					
Wk 52, female rat (Klaunig et al., 2015)		7				−15					−10			
Wk 52, male rat (Klaunig et al., 2015)	194			2					27					

* Indicates instances where statistical significance ($p < 0.05$) compared to controls was reported by study authors; shaded cells represent doses not included in the individual studies.

Table 3-6. Percent change in aspartate aminotransferase due to PFHxA exposure in short-term, subchronic, and chronic oral toxicity studies

Study design and reference	Dose (mg/kg-d)													
	2.5	5	10	15	20	30	50	62.5	100	125	200	250	500	1,000
28-d, female rat (NTP, 2018)								−1		0		0	11*	18*
28-d, male rat (NTP, 2018)								3		1		6	16*	36*
90-d, female rat (Chengelis et al., 2009b)			38				18				5			
90-d, male rat (Chengelis et al., 2009b)			−3				16				9*			
90-d, female rat (Loveless et al., 2009)					−58				−44				−36	
90-d, male rat (Loveless et al., 2009)					74				25*				39*	
Wk 26, female rat (Klaunig et al., 2015)		−10				−64					−65			
Wk 26, male rat (Klaunig et al., 2015)	−4			−2					−63					
Wk 52, female rat (Klaunig et al., 2015)		11				−11					−15			
Wk 52, male rat (Klaunig et al., 2015)	32			−1					13					

* Indicates instances where statistical significance ($p < 0.05$) compared to controls was reported by study authors; shaded cells represent doses not included in the individual studies.

Table 3-7. Percent change in alkaline phosphatase due to PFHxA exposure in short-term, subchronic, and chronic oral toxicity studies

Study design and reference	Dose (mg/kg-d)													
	2.5	5	10	15	20	30	50	62.5	100	125	200	250	500	1,000
28-d, female rat (NTP, 2018)								8		19		2	7	38*
28-d, male rat (NTP, 2018)								−4		−2		2	22*	51*
90-d, female rat (Chengelis et al., 2009b)			−5				−22				4			
90-d, male rat (Chengelis et al., 2009b)			−2				15				34*			
90-d, female rat (Loveless et al., 2009)					−16				24				−18	
90-d, male rat (Loveless et al., 2009)					17				20				60*	
Wk 26, female rat (Klaunig et al., 2015)		16				27					7			
Wk 26, male rat (Klaunig et al., 2015)	−4			1					−1					
Wk 52, female rat (Klaunig et al., 2015)		−18				4					−12			
Wk 52, male rat (Klaunig et al., 2015)	−15			−5					−2					

* Indicates instances where statistical significance ($p < 0.05$) compared to controls was reported by study authors; shaded cells represent doses not included in the individual studies.

Blood Proteins

The two major classes of proteins in the blood stream, albumin, and globulin, are made by the liver (with some globulins also made by the immune system) (Boron and Boulpaep, 2017). Blood proteins are routinely measured in diagnostic panels because changes in blood protein levels, particularly a decrease, can be indicators of protein loss due to kidney disease or impeded production in the liver, such as in liver disease (Boron and Boulpaep, 2017). Blood protein measures (total protein and globulin) were, in general, decreased across short-term (NTP, 2018), and subchronic (Chengelis et al., 2009b; Loveless et al., 2009) studies, with consistent and coherent dose-dependent findings across study designs. No PFHxA-related treatment effects on blood proteins were found in the chronic study at doses up to 100 or 200 mg/kg-day PFHxA (the highest doses administered) in male or female rats, respectively. The pattern of findings suggests a primary effect on blood globulins (decreased) in response to PFHxA exposure that was driving decreases in total protein and increases in the albumin:globulin ratio (A:G). These findings are discussed below and detailed information can be viewed in Figure 3-4 or by clicking on the HAWC link.

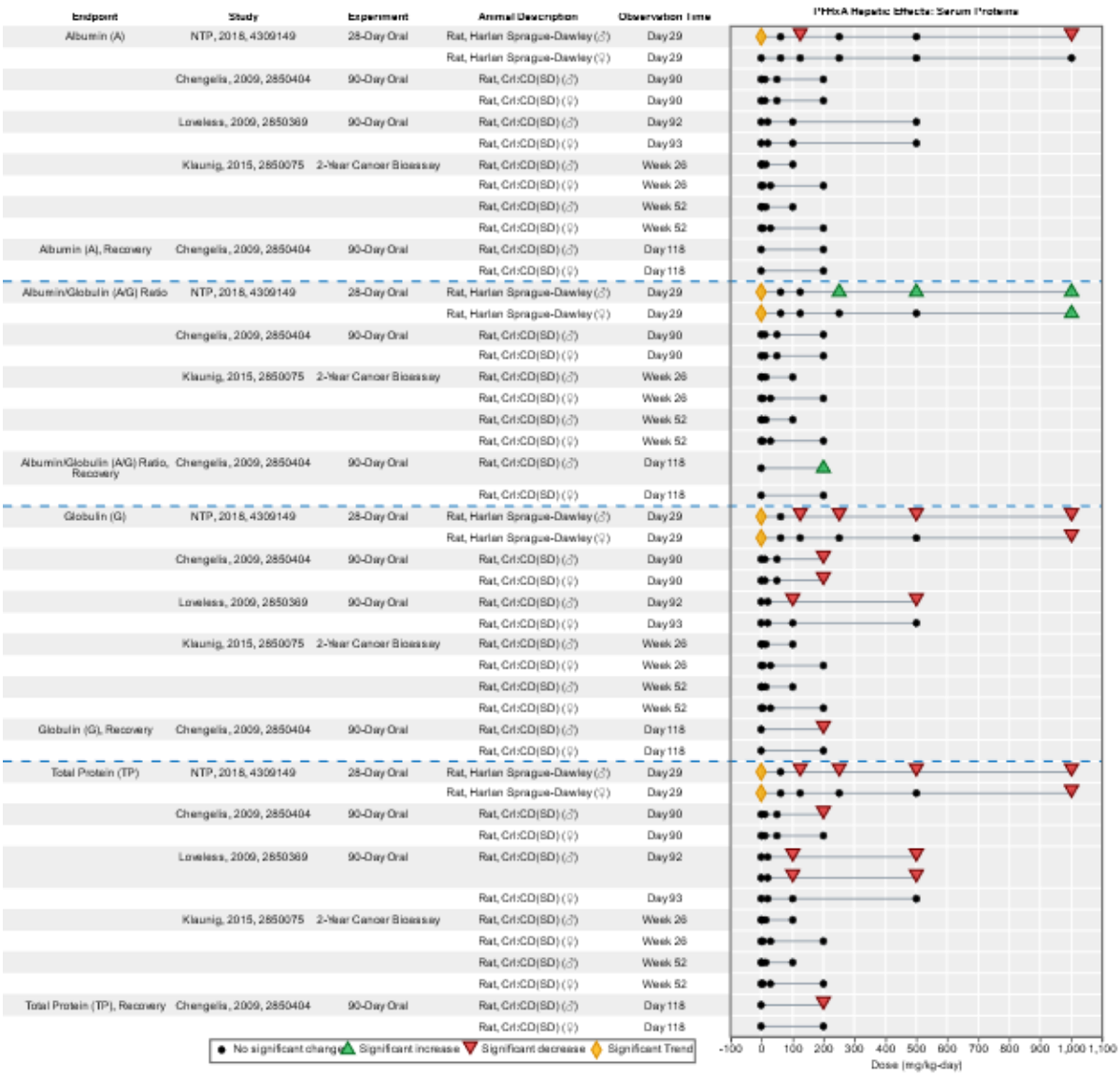


Figure 3-4. Blood protein findings after short-term, subchronic, and chronic PFHxA exposures (full details available by clicking the HAWC link).

Effects on total protein (TP; see Table 3-8)—the total amount of albumin and globulin found in blood, is associated with chronic liver disease (Whalan, 2015)—was decreased up to 20% in male rats receiving a dose ≥125 mg/kg-day in the 28-day study (with a significant trend) (NTP, 2018). A dose-responsive decrease (6%–14%, ≥100 mg/kg-day) in TP also was observed in male rats (Chengelis et al., 2009b; Loveless et al., 2009) with decreased levels observed in males (−6%, 200 mg/kg-day) at the 30-day recovery (Chengelis et al., 2009b). Albumin is a major blood protein that binds fatty acids, cations, bilirubin, thyroxine (T4), and other compounds. Decreased albumin levels are associated with decreased synthesis in the liver, increased catabolism, severe diffuse liver disease, subacute hepatitis, hepatocellular damage, ascites, cirrhosis, and chronic alcoholism (Whalan, 2015). Slight decreases ($p < 0.05$) in albumin were reported only in males exposed for

28 days to 125 mg/kg-day (6% decrease) and 1,000 mg/kg-day (7% decrease) PFHxA (NTP, 2018). The biological significance of this magnitude of change is unclear. No effects on albumin were identified in the subchronic or chronic studies.

Globulin, a collection of blood proteins larger than albumin made by both the liver and immune system, were decreased in all but the chronic study (see Table 3-9). Globulin decreases were observed in both male and female rats treated with PFHxA in the short-term study at ≥125 mg/kg-day and 1,000 mg/kg-day, respectively (NTP, 2018). Consistent with the short-term study, decreases were also observed in both males and females in the highest dose groups [200 (Chengelis et al., 2009b) and 100 mg/kg-day (Loveless et al., 2009)]. Notably, globulin decreases (10%) persisted at the 30-day recovery in males (200 mg/kg-day) and returned to normal in females (Chengelis et al., 2009b).

The decrease in globulin was consistent with increases in A:G, a routine blood test used to screen for liver, kidney, immune, and gastrointestinal function. The A:G was increased in males and females (113%–160% at ≥250 mg/kg-day and 142% at 1,000 mg/kg-day) with significant trends in both sexes (NTP, 2018). Chengelis et al. (2009b) observed an increase (10%) at the 30-day recovery in rats receiving an oral dose of 200 mg/kg-day.

Table 3-8. Percent change in total protein due to PFHxA exposure in short-term, subchronic, and chronic oral toxicity studies

Study design and reference	Dose (mg/kg-d)													
	2.5	5	10	15	20	30	50	62.5	100	125	200	250	500	1,000
28-d, female rat (NTP, 2018)								0		1		−1	−1	−7*
28-d, male rat (NTP, 2018)								−4		−7*		−7*	−10*	−20*
90-d, female rat (Chengelis et al., 2009b)			4				3				−4			
90-d, male rat (Chengelis et al., 2009b)			−3				−4				−6*			
90-d, female rat 30-d recovery (Chengelis et al., 2009b)											−3			
90-d, male rat 30-d recovery (Chengelis et al., 2009b)											−6*			
90-d, female rat (Loveless et al., 2009)					−1				−1				−3	
90-d, male rat (Loveless et al., 2009)					0				−6*				−14*	

Study design and reference	Dose (mg/kg-d)													
	2.5	5	10	15	20	30	50	62.5	100	125	200	250	500	1,000
2-yr, female rat (Klaunig et al., 2015)		−1				1					0			
2-yr, male rat (Klaunig et al., 2015)	−1			0					−3					

* Indicates instances where statistical significance ($p < 0.05$) compared to controls was reported by study authors; shaded cells represent doses not included in the individual studies.

Table 3-9. Percent change in globulins due to PFHxA exposure in short term, subchronic, and chronic oral toxicity studies

Study design and reference	Dose (mg/kg-d)													
	2.5	5	10	15	20	30	50	62.5	100	125	200	250	500	1,000
28-d, female rat (NTP, 2018)								−6		−4		−9	−7*	−28*
28-d, male rat (NTP, 2018)								−7		−9		−14*	−24*	−40*
90-d, female rat (Chengelis et al., 2009b)			0				0				−15*			
90-d, male rat (Chengelis et al., 2009b)			−7				−11*				−11*			
90-d, female rat 30-d recovery (Chengelis et al., 2009b)											0			
90-d, male rat 30-d recovery (Chengelis et al., 2009b)											−10*			
90-d, female rat (Loveless et al., 2009)					0				−3				−11*	
90-d, male rat (Loveless et al., 2009)					0				−13*				−28*	
2-yr, female rat (Klaunig et al., 2015)		−4				4					−4*			
2-yr, male rat (Klaunig et al., 2015)	−4			4					−4					

* Indicates instances where statistical significance ($p < 0.05$) compared to controls was reported by study authors; shaded cells represent doses not included in the individual studies.

Hepatobiliary Components

Other indicators of potential liver dysfunction or injury included impacts on bile components essential for normal lipid metabolism and red blood cell breakdown. ALP (discussed with serum enzymes and in Table 3-7, see above) is an indicator of bile duct obstruction and was consistently increased in male and female rats in the short-term study (NTP, 2018) and subchronic studies (Chengelis et al., 2009b; Loveless et al., 2009). In the short-term study (NTP, 2018), bile acids were increased at the highest dose (1,000 mg/kg-day) with a significant trend (a possible indication of cholestatic liver injury), and bilirubin was decreased in a dose-response gradient across both the short-term and subchronic (Loveless et al., 2009) studies (see Figure 3-5). Lower than normal bilirubin levels are usually not a concern and can be reduced in response to increased conjugation rates after hepatic enzyme induction and excretion into bile (Hall et al., 2012).

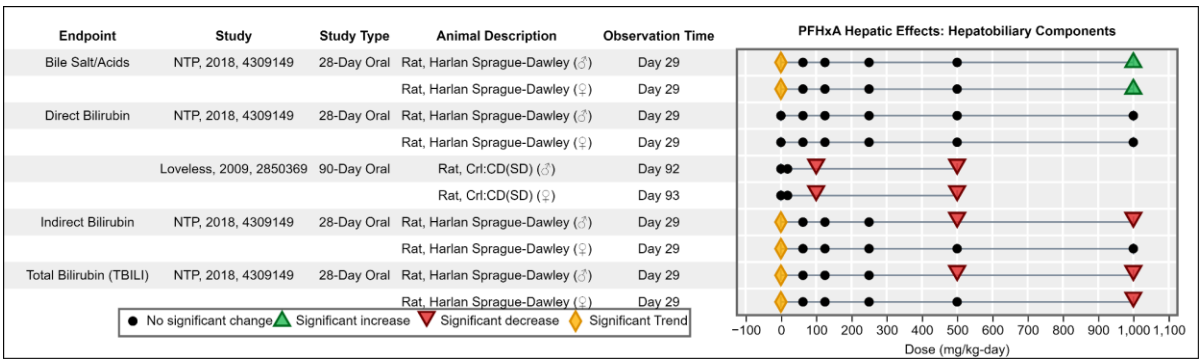


Figure 3-5. Hepatobiliary findings in rats exposed by gavage to PFHxA or PFHxA sodium salt (full details available by clicking the HAWC link).

Mechanistic Evidence and Supplemental Information

The available evidence base reports increased liver weight, hepatocellular hypertrophy, hepatocellular necrosis, increased (1.5–2.5-fold) serum enzymes, increased peroxisomal beta oxidation, decreased total protein (driven by decreased globulin), and decreased bilirubin levels in rats exposed to PFHxA. This collection of findings was considered for their potential adversity and relevance in humans using Hall criteria (Hall et al., 2012) along with other available supplemental mechanistic evidence. The mechanistic findings (i.e., -omics data, transactivation assays, oxidative stress assays, knockout mouse models) were coherent with animal findings of PFHxA mediated liver injury. The supplemental mechanistic evidence from other short-chain PFAS (e.g., PFBA) supported both PPARα independent and dependent pathways that are relevant in humans, increasing the plausibility that these PFAS-mediated responses also apply to PFHxA. Additional evidence from non-mammalian models is available and summarized in the PFHxA Literature Tagtree.

Considerations for Potentially Adaptive Versus Adverse Responses

Considering that the hepatic effects of PFHxA exposure (increased liver weight and hepatocyte hypertrophy) observed in rodents could represent adaptive responses, the potential adversity of these effects was a key consideration and analyzed. In the absence of a known mechanism leading to increased liver weight, hepatocellular hypertrophy, and necrosis, the evidence on PFHxA-mediated hepatotoxicity was evaluated to inform interpretations regarding adversity utilizing guidance from Hall et al. (2012). Increased liver weight and hepatocellular hypertrophy can be associated with changes that are adaptive in nature (Hall et al., 2012) and not necessarily indicative of adverse effects unless observed in concordance with other clinical, pathological markers of overt liver toxicity (see Appendix A). The IRIS PFAS Assessment Protocol (which applies to PFHxA) states the panel recommendations from Hall et al. (2012) can be used to judge whether observed hepatic effects are adverse or adaptive in nature. Given that Hall et al. (2012) was focused on framing noncancer liver effects in the context of progression to liver tumors, however, the protocol further indicates that "…consultation of additional relevant information will be considered to interpret the adversity of noncancer liver effects over a lifetime exposure, taking into account that effects perceived as adaptive can progress into more severe responses and lead to cell injury." Hall et al. (2012) indicates that additional evidence must be considered when determining adverse versus adaptive response in rodents in the context of hepatocellular hypertrophy that include:

1) Is there histological evidence of the following structural degenerative or necrotic changes?
    - Hepatocyte necrosis, fibrosis, inflammation, and steatotic vacuolar degeneration
    - Biliary/oval cell proliferation, degeneration, fibrosis, and cholestasis
    - Necrosis and degeneration of other resident cells within the liver
2) In the absence of histological changes, using a weight-of-evidence approach, is there clinical pathology evidence of hepatocyte damage characterized by a dose dependent and biologically significant and consistent increase in at least *two* of the following liver parameters?
    - At least ×2 to ×3 increase in ALT
    - A biologically significant change in other biomarkers of hepatobiliary change (ALP, AST, γGT, GLDH, etc.)
    - A biologically significant change in another clinical pathology marker indicating liver dysfunction (albumin, bilirubin, bile acids, coagulation factors, cholesterol, triglycerides, etc.)

With regard to Step 1 above, histological evidence of structural change included necrosis in females rats only (incidence of 12/70) receiving 200 mg/kg-day in the chronic study (note the

highest dose in male rats was half the female dose, 100 mg/kg-d)(Klaunig et al., 2015) and 2.5-5 times lower than the highest dose in the short term (NTP, 2018) or the subchronic studies (Chengelis et al., 2009b; Loveless et al., 2009). There was one incidence of necrosis male rats only (n = 1/10 vs. 0/10 in controls, not significant) from the short-term study (NTP, 2018), and no findings of necrosis from either subchronic study (Chengelis et al., 2009b; Loveless et al., 2009). Histological findings did include increased incidence of hepatocellular hypertrophy from the short term and both subchronic studies. Notably, hypertrophy findings persisted in both male and female rats 90-day after recovery (Loveless et al., 2009). Although some uncertainties remain, the necrosis observed in the female rats in the chronic study (and at a highest dose in a short-term study) support the adversity of the hepatic effects of PFHxA regarding Step 1.

Regarding Step 2 above, other liver parameter effects were observed after PFHxA exposure and included increased peroxisomal beta oxidation in both subchronic studies (Chengelis et al., 2009b; Loveless et al., 2009) that persisted at 30 days recovery in both male and female rats (Loveless et al., 2009). The serum enzymes ALT was increased 2-3-fold in the short term and subchronic studies. AST and ALP were also significantly increased, although the magnitude of the response was <2-fold, at the same or lower doses than the observed increases in hepatocellular hypertrophy. Other parameters characterized by a dose-dependent PFHxA-mediated effect included decreased globulin, decreased total protein, and decreased bilirubin. While the collection of findings was generally observed across both sexes, the magnitude of the change was greater in males than females. This sex-specific difference is possibly explained by the increased clearance rate in females compared with males. Although some uncertainties remain, the observed liver parameter changes in the female and male rats in the subchronic studies (e.g., increased liver enzymes, increased peroxisomal beta oxidation, decreased blood proteins) support the adversity of the hepatic effects of PFHxA regarding Step 1.

Considering the Hall et al. (2012) criteria above, the observed increase in relative liver weight and hepatocellular hypertrophy in rats exposed to PFHxA are interpreted as adverse, human relevant, and potentially leading to increasingly severe outcomes such as necrosis.

Peroxisomal Beta Oxidation

Peroxisomal proliferation can be induced within the peroxisomes to perform beta oxidation of lipids into acetyl CoA and hydrogen peroxide (H_2O_2) (Reddy, 2004). Hydrogen peroxide is a reactive metabolite and can cause oxidative damage to the surrounding tissue. Two subchronic studies measured PFHxA induction of peroxisomal beta oxidation activity in male and female rats (Chengelis et al., 2009b; Loveless et al., 2009) (see Figure 3-6) and both were considered *medium* or *high* confidence for this outcome. Chengelis et al. (2009b) reported an increase (p < 0.05, 1.37-fold) in males treated with 200 mg/kg-day at 13 weeks. Loveless et al. (2009) found increased peroxisomal beta oxidation activity in both sexes gavaged with 500 mg/kg-day for 10 and 90 days (males, 3.1- and 4.36-fold, respectively; females, 1.45- and 2.67-fold, respectively). Notably, increased activity persisted after the 30-day recovery and male rats were more sensitive than

females, with males in the 100 mg/kg-day group also showing increased peroxisomal beta oxidation (Loveless et al., 2009).

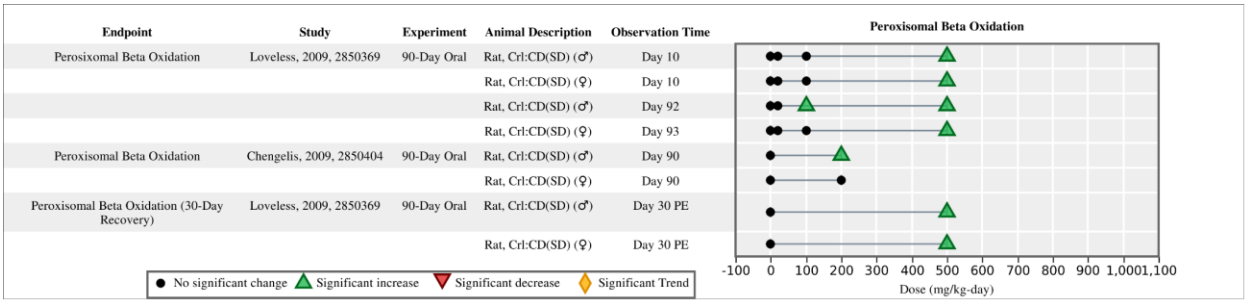


Figure 3-6. Peroxisomal beta oxidation activity in rats exposed by gavage to PFHxA or PFHxA sodium salt (full details available by clicking the HAWC link).

Considerations Related to Human Relevance

Peroxisomal beta oxidation in rodents is mediated primarily by PPARα. The induction of both PPARα and CAR target gene expression were observed after PFHxA exposure in both the short-term and subchronic rodent studies (NTP, 2018; Chengelis et al., 2009b; Loveless et al., 2009). Specifically, in the short-term study, NTP (2018), in vivo PFHxA exposure in rats elicited significant dose-related increases in the liver expression of the PPARα target genes acyl-CoA oxidase (*Acox1*, up to 2-fold increase) and cytochrome P450 4a1 (*Cyp4a1*, up to 12.5-fold increase). In the same short-term study, constitutive androstane receptor (CAR) target genes cytochrome P450 2b1 (*Cyp2b1*, up to 7-fold increase) and cytochrome P450 2b2 (*Cyp2b2*, up to 3-fold increase) were also induced after PFHxA exposure. Functional evidence of PPARα activation by PFHxA exposure was provided by the NTP (2018) short-term study where increases (up to 16-fold) in Acyl-CoA oxidase activity in male rats receiving >250 mg/kg-day PFHxA (not measured in females) were observed.

The hepatic effects of PFHxA exposure observed in the rat studies discussed above were also evaluated in pathogen-free ICR mice receiving 0, 50, or 200 mg/kg PFHxA by oral gavage daily for 60 days (Jiang et al., 2021). The observed hepatotoxicity in ICR mice was similar to the SD rats and included histopathological findings of hepatocellular hypertrophy, inflammatory cell infiltration, and degeneration. These findings were coherent with results from whole liver RNA-seq and proteomics analysis that identified pathways enriched with differentially regulated genes and proteins involved in PPAR signaling pathways (Jiang et al., 2021).

The available evidence also included evidence for mechanistic information supporting biologically plausible pathways (including those mediated by PPARα activation) leading to the observed PFHxA-mediated hepatoxicity in rodents and considered relevant in humans. For example, PFHxA activation of both rodent and human PPARα in HepG2 cells at concentrations between 1–30 μM; 12.2 μM PFHxA Buhrke et al. (2013). The activation of human PPARα by short and long chain PFAS was also observed by Behr et al. (2020) in human HepG2 cells. In another

study, Wolf et al. (2008) examined, PPARα activation by PFHxA in COS1 cells transfected with reporter gene constructs containing either the mouse or human PPARα ligand binding domain fused to a Gal4 DNA binding domain under control of an SV-40 promoter in a luciferase reporter plasmid. These assays indicated that both mouse and human PPARα are activated by PFHxA in a treatment-related manner with PFHxA being a more potent activator of the human (lowest-observed-effect concentration, LOEC = 10 μM) compared to the mouse (LOEC = 20 μM) receptor (Wolf et al., 2008). While the transactivation studies of Wolf et al. (2008) indicated PFHxA activation of both the mouse and human PPARα, significant effects were reported only for treated vs. control within a species. In sharp contrast to all other studies Buhrke et al. (2013), Wolf et al. (2008), Wolf et al. (2008), and Jiang et al. (2021), Bjork and Wallace (2009) reported PFAS activation of PPARα target gene expression in rat, but not human immortalized and primary hepatocytes exposed. However, the outcome was considered low confidence due to notable concern that variables were unaccounted or uncontrolled for (i.e., no positive control, no baseline characterization of gene expression changes or PPAR activity).

Additional in vitro evidence for PFHxA effects in human cell lines (including HepG2 and HepaRG cells) is available from EPA's CompTox Chemicals Dashboard (U.S. EPA, 2018a). Transactivation assays in immortalized human HepG2 cells indicated PFHxA treatment effects that included activation of the transcription factors PPARα and hypoxia inducible factor 1 subunit alpha (HIF1α, a transcriptional regulator of genes involved in the hypoxia response). Similarly, gene expression assays in human-derived HepaRG cells identified the induction of 16 genes including several cytochrome P450 family members, transporters, kinases, and oxidase/oxidoreductase related activities that are primarily involved in oxidation/reduction and/or lipid metabolism (see Table 3-10), consistent with evidence of short and long chain PFAS (including PFHxA) activation of both human and rodent PPAR-dependent and independent pathways. Altogether, the mostly consistent activation across species of PPARα by PFHxA provides further support for the human relevance of the PFHxA-induced liver changes observed in rodents.

Table 3-10. Gene targets identified from EPA CompTox Chemicals Dashboard after PFHxA treatment in human liver cell lines[a]

Gene Symbol	Gene Name	AC50[b]	LOGAC50	BMAD[c]
*ABCG2**	ATP-binding cassette, sub-family G (WHITE), member 2 (Junior blood group)	9.49	0.977	0.201
*ACOX1**	acyl-CoA oxidase 1, palmitoyl	9.47	0.976	0.135
ADK	adenosine kinase	2.88	0.459	0.166
*CYP2B6**	cytochrome P450, family 2, subfamily B, polypeptide 6	19.1	1.28	0.251
CYP2C19	cytochrome P450, family 2, subfamily C, polypeptide 19	5.21	0.717	0.187
*CYP2C8**	cytochrome P450, family 2, subfamily C, polypeptide 8	7.88	0.896	0.216
CYP2C9	cytochrome P450, family 2, subfamily C, polypeptide 9	6.53	0.815	0.314

Gene Symbol	Gene Name	AC50[b]	LOGAC50	BMAD[c]
CYP3A7	cytochrome P450, family 3, subfamily A, polypeptide 7	10.3	1.01	0.259
CYP4A11	cytochrome P450, family 4, subfamily A, polypeptide 11	45.3	1.66	0.213
CYP4A22	cytochrome P450, family 4, subfamily A, polypeptide 22	57.6	1.76	0.269
*FABP1**	fatty acid binding protein 1, liver	22.3	1.35	0.161
FMO3	flavin containing monooxygenase 3	13.3	1.12	0.145
*HMGCS2**	3-hydroxy-3-methylglutaryl-CoA synthase 2 (mitochondrial)	4.13	0.616	0.175
*PDK4**	pyruvate dehydrogenase kinase, isozyme 4	20.9	1.32	0.161
SLCO1B1	solute carrier organic anion transporter family, member 1B1	9.82	0.992	0.181
UGT1A1	UDP glucuronosyltransferase 1 family, polypeptide A1	15.1	1.18	0.221

* PPARα target gene (http://www.ppargene.org/index.php).
[a]After filtering out results flagged in the Dashboard as uncertain (e.g., high degree of variability) or occurring at high concentrations associated with cytotoxicity (U.S. EPA, 2018c; Filer et al., 2016; Filer, 2015), 19 assay targets remained from human liver cell-based assays (at up to 200 µM PFHxA).
[b]AC50 – active concentration (µM) that elicited half maximal response.
[c]BMAD – baseline median absolute deviation.

The data discussed above suggest similar PPARα activation occurs in both rodents and humans (at least in vitro). Potential pathways such as PPARα and CAR activation can contribute to some of the hepatic changes caused by PFHxA exposure, including hypertrophy. Studies of the prototypical PPARα agonist, WY-14643, indicate an increased sensitivity of rodents as compared to humans; however, the PFHxA-specific data do not demonstrate such clear differences with this structurally different compound.

Overall, although the PFHxA-specific data informing possible biological pathways leading to the observed hepatic effects are sparse and uncertainties remain, collectively the available in vivo and in vitro evidence for PFHxA support the potential for the responses observed in rodents to be relevant to humans.

Evidence from Other PFAS

Although no direct in vivo evidence is available for PFHxA effects in PPARα null rodent models, evidence from other PFAS, as well as PFAS exposures in PPARα null and humanized mouse models provide evidence that PPARα independent and dependent pathways from other PFAS are consistent for other short-chain PFAS (e.g., PFBA) (U.S. EPA, 2022b), and possibly long-chain PFAS as well (e.g., PFNA, PFOA, PFOS) (U.S. EPA, 2023a, b), supporting the plausibility that these PFAS-mediated responses also apply to PFHxA. Specifically. Specifically, Foreman et al. (2009) also observed increased liver weight, hepatic lipid accumulation, ALT increases >2-fold, and pathologically similar (severity and incidence) hepatocellular hypertrophy in male SV129 wild-type SV and humanized PPARα mice exposed to PFBA. Although there is no evidence specifically challenging the role of PPARα in PFHxA-mediated hepatotoxicity, based on PFHxA structural

similarity with other PFAS, most notably PFBA, it is reasonable to infer that PFHxA exposure in genetic mouse model systems would elicit similar effects.

Evidence Integration

The human evidence base is limited to a single *medium* confidence study reporting null associations between serum biomarker levels and PFHxA exposure. Based on these data, there is *indeterminate* human evidence of hepatic effects.

The hepatic findings in rodents exposed to PFHxA included increased relative liver weight observed with increased hepatocellular hypertrophy at doses as low as 100 mg/kg-day (Loveless et al., 2009) and 200 mg/kg-day (Chengelis et al., 2009b) in male rats that persisted after 30- and 90-day recovery. Corroborative evidence for adverse hepatotoxicity included increased serum enzymes, (e.g., ALT increased >2-fold) in the subchronic studies, although a dose-responsive relationship was observed in the short term, but not the subchronic, studies. Serum enzyme changes were not observed in the chronic study (Klaunig et al., 2015). Hepatocellular necrosis was observed in male rats in a *high* confidence short term study (NTP, 2018) at 1,000 mg/kg-day, *low* confidence subchronic study (Chengelis et al., 2009b) and in the *high* confidence chronic study (female rats) (Klaunig et al., 2015) at 200 mg/kg-day. Other clinical findings altered by PFHxA exposure included decreased bilirubin and decreased total protein mainly driven by decreases in globulins (see Clinical Chemistry section above). These findings (i.e., increased liver weight with concurrent hepatocellular hypertrophy, increases in ALT, and decreased protein levels) were considered adverse as they might lead to the necrosis observed in females at 200 mg/kg-day in the chronic study. In general, the pattern of findings suggests a generally increased sensitivity in males as compared to females. Overall, there is *robust* animal evidence of hepatic effects. This judgment is based on four studies in Sprague-Dawley rats that were generally rated *high* confidence on the outcome-specific evaluations.

Although multiple biological pathways could lead to the histopathological findings mentioned above, the PFHxA database for molecular evidence was predominantly limited to PPARα pathways and included in vitro assays measuring PFHxA induction of PPARα activity (Wolf et al., 2014; Wolf et al., 2008), peroxisomal beta oxidation activity (NTP, 2018; Chengelis et al., 2009b; Loveless et al., 2009), changes in gene expression for CAR and PPARα cytochrome P450 gene expression (NTP, 2018), and in vivo PPARα knockout and humanized genetic mouse models exposed to PFAS structurally similar to PFHxA (Das et al., 2017; Rosen et al., 2017; Foreman et al., 2009). Wolf et al. (2008) and Wolf et al. (2014) found evidence that PFHxA activated the human PPARα receptor concentrations that in rodent. Dose-responsive increases in peroxisomal beta oxidation activity were observed in two subchronic studies (Chengelis et al., 2009b; Loveless et al., 2009) at a dose as low as 100 mg/kg-day and this effect persisted after the 30-day recovery (Loveless et al., 2009). Evidence for pathways other than PPARα and CAR were available from genetic PPARα knockout mouse studies evaluating the effects of PFAS exposure (Das et al., 2017; Rosen et al., 2017; Foreman et al., 2009) that found similar levels of increased liver weight and

incidence of hepatocellular hypertrophy when comparing between PPARα knockout, humanized, and wild-type mouse models, suggesting PPARα dependent and independent pathways are activated by other PFAS, most notably PFBA (U.S. EPA, 2022b), a short-chain PFAS like PFHxA. Further supplemental mechanistic evidence indicated that human PPARα and related gene expression is activated by PFHxA treatment in vitro. Thus, taken together, the findings observed in rodent models are interpreted as relevant to humans.

Overall, the currently available ***evidence indicates*** that PFHxA likely causes hepatic effects in humans given sufficient exposure conditions (see Table 3-11).[5] This conclusion is based on studies of animals showing increased liver weight, hepatocellular hypertrophy, increased serum enzymes (including >2-fold ALT) and decreased serum globulins generally occurring at ≥100 mg/kg-day within the evidence base of four primarily *high* confidence studies of short-term, subchronic, and chronic PFHxA exposure in (primarily male) rats. The findings in rats were determined to be adverse and relevant to humans, with the likely involvement of both PPARα-dependent and -independent pathways and consistent with hepatic effects identified for other PFAS (U.S. Environmental Protection Agency: (U.S. EPA, 2022b; ATSDR, 2021; U.S. EPA, 2021a, 2018b, 2016a, b)) (summarized in Section 4, see Table 4-2).

[5] The "sufficient exposure conditions" are more fully evaluated and defined for the identified health effects through dose-response analysis in Section 5.

Table 3-11. Evidence profile table for hepatic effects

<table>
<tr><th colspan="5">Evidence stream summary and interpretation</th><th>Evidence integration summary judgment</th></tr>
<tr><td colspan="5">Evidence from studies of exposed humans</td><td rowspan="6">⊕⊕⊙
Evidence indicates (likely)

Primary basis:
Four generally high confidence studies in rats ranging from short-term to chronic exposure, generally in males at ≥100 mg/kg-d PFHxA

Human relevance:
Given the induction of human PPARα by PFHxA, as well as support for involvement of both PPARα-dependent and independent pathways for PFHxA and related PFAS, effects in rats are considered relevant to humans

Cross-stream coherence:
N/A (human evidence indeterminate)

Susceptible populations and lifestages:
No evidence to inform</td></tr>
<tr><td>Studies and confidence</td><td>Factors that increase certainty</td><td>Factors that decrease certainty</td><td>Summary and key findings</td><td>Evidence stream judgment</td></tr>
<tr><td><u>Serum Biomarkers</u>
2 <u>low</u> confidence studies</td><td>• No factors noted</td><td>• Low confidence studies (low sensitivity)</td><td>• No association of PFHxA with serum biomarkers</td><td>⊙⊙⊙
Indeterminate</td></tr>
<tr><td colspan="5">Evidence from animal studies</td></tr>
<tr><td>Studies and confidence</td><td>Factors that increase certainty</td><td>Factors that decrease certainty</td><td>Summary and key findings</td><td>Evidence stream judgment</td></tr>
<tr><td><u>Organ Weight</u>
3 <u>high</u> confidence:
28-d
90-d (2 studies)</td><td>• Consistent increases, all studies, and sexes
• Dose-response observed in all studies
• Coherence with histopathology
• Large magnitude of effect, up to 63%
• High confidence studies</td><td>• No factors noted</td><td>• Increased liver weight at ≥200 mg/kg-d; stronger in males</td><td>⊕⊕⊙
Moderate

Findings considered adverse based on observed liver parameter changes in the female and male rats in the subchronic studies (e.g., increased liver enzymes, increased peroxisomal beta oxidation, decreased blood proteins) at similar doses, but lower than doses at which necrosis was observed (200 mg/kg-d) consistent with the progressive change to necrosis support the adversity of</td></tr>
<tr><td><u>Histopathology</u>
3 <u>high</u> confidence studies in adult rats:
28-d
90-d
2-yr</td><td>• Consistent cellular hypertrophy across studies and sexes
• Coherence with liver weight</td><td>• No factors noted</td><td>• Cellular hypertrophy at ≥100 mg/kg-d; stronger in males
• Necrosis in females at</td></tr>
</table>

Evidence stream summary and interpretation					Evidence integration summary judgment
1 *low* confidence study in adult rats: 90-d	• *Dose-response* for hypertrophy • Concerning *severity of effect*—necrosis (with short term, subchronic, and chronic exposure) • *High* confidence studies		200 mg/kg-d	the hepatic effects with potential for progression for more severe phenotypes, including necrosis with longer-term exposure.	
Serum Biomarkers of Hepatic Injury 4 *high* confidence studies in adult rats: 28-d 90-d (2 studies) 2-yr	• *Consistent* increases in ALT, AST, and ALP, and decreases in bilirubin, across studies • *Magnitude of effect*, >2-fold ALT • *Dose-response* for total protein • *Coherence* of ALP and bilirubin • *High* confidence studies	• No factors noted	• Increased ALT, AST, ALP, and bile salts/acids at ≥20, ≥100, ≥200, and 500 mg/kg-d, respectively; stronger in males • Decreased total protein at ≥100 mg/kg-d; stronger in males		
Mechanistic evidence and supplemental information					
Biological events or pathways	**Primary evidence evaluated** **Key findings, interpretation, and limitations**			**Evidence stream summary**	
Molecular Initiating Events—PPARα	*Key findings and interpretation:* • Consistent studies demonstrating In vitro induction of human and rodent PPARα activity in trans-activation experiments. Reporter gene activation at lower effective concentrations in human versus mouse constructs.			• Biologically plausible support for PPARα-dependent and independent pathways contributing to hepatic	

Evidence stream summary and interpretation			Evidence integration summary judgment
	• Induction of PPARα in association with hepatic effects in a short-term oral exposure study. • PFHxA binding to and activation of human PPARα. • Activation of PPARα target gene expression in rodents. • Findings coherent with animal findings of increased peroxisomal beta oxidation activity.	effects of PFHxA that are relevant in humans.	
Molecular Initiating Events—Other Pathways	*Key findings and interpretation*: • Indirect evidence supporting activation of PPARα-independent pathways contributing to hepatic effects like those observed for PFHxA in PPARα knockout and humanized mice after short-term oral exposure to PFAS other than PFHxA, including the related PFAS, PFBA. • -omics evidence supporting differential regulation of genes regulated by PPARα dependent and independent pathways *Limitations*: Small evidence base with no experiments specifically challenging the role of PPARα in PFHxA-induced hepatic injury.		
Organ Level Effects	*Key findings and interpretation*: • Increased peroxisomal beta oxidation activity that persisted 30 d post-exposure (likely not a transient, adaptive response) in short-term and subchronic rat studies of oral PFHxA exposure. • Indirect evidence of fatty liver, hepatocellular hypertrophy, and hepatomegaly in PPARα KO mice after short-term oral exposure to PFAS other than PFHxA. *Limitations*: Small evidence base and the most compelling in vivo evidence for PPARα-independent pathways with hepatic effects did not specifically test PFHxA.		

3.2.2. Developmental Effects

Human

A single study evaluated associations between PFHxA and developmental effects, specifically birth size and postnatal growth, in a study in China with follow-up to 5 months after birth (Jin et al., 2020). This study was considered *uninformative* and not considered further due to lack of consideration of potential confounding, including parity, socioeconomic status and maternal age.

Animal

Three studies described in two publications examined developmental outcomes, including offspring viability, body weight, eye opening, and malformations and variations. Rats were exposed to PFHxA sodium salt during gestation (gestation day [GD] 6–20; developmental study) or continuously exposed throughout gestation and lactation (reproductive study) (Loveless et al., 2009). Mice were exposed to PFHxA ammonium salt from GD 6–18 (Iwai and Hoberman, 2014). These studies were rated *high* confidence. The results from outcome-specific, confidence evaluations for all individual reproductive outcomes are presented in Table 3-12, and details are available by clicking the HAWC link. Effects on male and female reproductive system development following developmental exposure are discussed in the male and female reproductive effects sections, respectively (see Sections 3.2.6 and 3.2.7).

Table 3-12. Study design characteristics and outcome-specific study confidence for developmental endpoints

Study	Species, strain (sex)	Exposure design	Exposure route and dose	Offspring viability	Offspring body weight	Eye Opening	Malformations and Variations
Loveless et al. (2009)	Rat, Crl:CD(SD) Sprague–Dawley (male and female)	Reproductive study: P_0 females dosed 70 d prior to cohabitation, through gestation and lactation (126 d); P_0 males dosed for 110 d Developmental study: GD 6–20	Gavage[a] Female: 0, 20, 100, 500 mg/kg-d	++	++	++	++
Iwai and Hoberman (2014)[c]	Mouse, Crl:CD1(ICR) (male and female)	Developmental: GD 6–18	Gavage[b] Phase 1: 0, 100, 350, 500 mg/kg-d Phase 2: 0, 7, 35, 175 mg/kg-d	++	++	++	NM

++ Outcome rating of high confidence; NM, outcome not measured.

[a,b]Study evaluation for animal toxicological developmental endpoints reported from studies with rats receiving PFHxA sodium salt[a] or PFHxA ammonium salt[b] by gavage. Study evaluation details for all outcomes are available by clicking the HAWC link.

[c]Phase 1 was a range finding study used to determine the appropriate dose range for identification of a NOAEL in Phase 2.

Offspring Mortality

Potential effects of PFHxA exposure on offspring viability were evaluated in a developmental study (Iwai and Hoberman, 2014) and a one-generation reproductive study (Loveless et al., 2009). Mice exposed to PFHxA ammonium salts during gestation (GD 6–18) showed a dose-dependent increase in the incidence of offspring mortalities occurring both pre- and postnatally (Iwai and Hoberman, 2014). Most deaths occurred between postnatal day (PND) 0–7, with a statistically significant increase observed in the 350 and 500 mg/kg-day dose groups on PND 1–4. These effects were observed across two experimental cohorts with different but overlapping dose ranges (cohort 1: 0, 100, 350, and 500 mg/kg-day; cohort 2: 0, 7, 35, and 175 mg/kg-day). Early postnatal losses are reflected in treatment-related effects on several measures of offspring viability for the 500 mg/kg-day dose group. Specifically, statistically significant changes were observed in the following related outcomes: decreased viability index for PND 0–4 and PND 0–7, fewer surviving pups per litter on PND 20, and increased incidence of total litter loss between PND 0–3 (5 of 17 for the 500 mg/kg-day group compared to 1 of 19 dams for concurrent controls). A dose-dependent increase in the number of stillbirths, a measure of prenatal mortality, was also reported across the two phases of the experiment (incidence of 3/241, 5/245, and 19/177 for the 175, 350, and 500 mg/kg-d dose groups, respectively). Results are summarized in Figure 3-7 and Table 3-13.

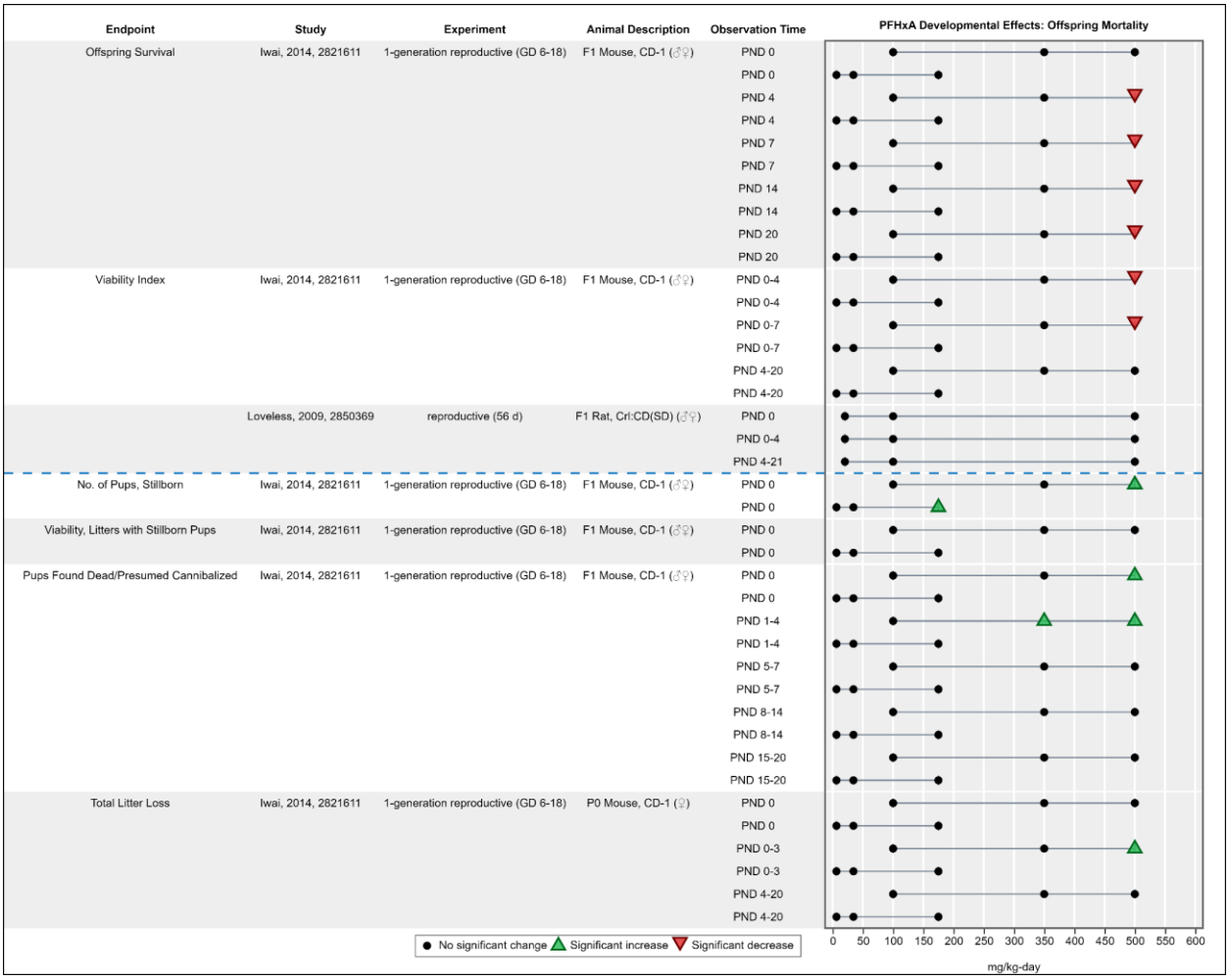


Figure 3-7. Developmental effects on offspring viability in mice exposed to PFHxA ammonium salt (HAWC: PFHxA – Animal Toxicity Developmental Effects link).

The Iwai and Hoberman (2014) study was conducted in two phases. Phase 1 was a range-finding study (100, 350, or 500 mg/kg-d) used to determine the appropriate doses (7, 35, 175 mg/kg-d) to identify a NOAEL in Phase 2.

Table 3-13. Incidence of perinatal mortality following PFHxA ammonium salt exposure in a developmental oral toxicity study

Study design and reference	Dose (mg/kg-d)							
	0 (Cohort 1)	0 (Cohort 2)	7	35	100	175	350	500
Stillbirths, male and female (combined) mice (Iwai and Hoberman, 2014)	4	0	0[a]	0	0	3*	5[a]	19[a]*
Mortalities, PND 0, male and female (combined) mice (Iwai and Hoberman, 2014)	0	0	0	0	0	4	3[a]	21[a]*

Study design and reference	Dose (mg/kg-d)							
	0 (Cohort 1)	0 (Cohort 2)	7	35	100	175	350	500
Mortalities, PNDs 1–4, male and female (combined) mice (Iwai and Hoberman, 2014)	2	1[a]	3[a]	2	2[a]	0[a]	13[a]*	15[a]*
Mortalities, PNDs 5–7, male and female (combined) mice (Iwai and Hoberman, 2014)	0[a]	1	0[a]	0	0[b]	3[a]	2[a]	0[a]
Mortalities, PNDs 8–14, male and female (combined) mice (Iwai and Hoberman, 2014)	0	0	0	0	0[a,b]	0[a]	0[a]	0[a]
Mortalities, PNDs 15–20, male and female (combined) mice (Iwai and Hoberman, 2014)	0	0	0	0	2[b]	1	0	0
Total pups delivered, male and female (combined) mice (Iwai and Hoberman, 2014)	221	249	211	232	250	241	245	177
Cumulative perinatal mortality/total pups delivered, male and female (combined) mice (Iwai and Hoberman, 2014)	6/ 221	2/ 249	3/ 211	2/ 232	4/ 250	11/ 241	23/ 245	55/ 177

The Iwai and Hoberman (2014) study was conducted in two phases. Phase 1 was a range-finding study (100, 350, or 500 mg/kg-d) used to determine the appropriate doses (7, 35, 175 mg/kg-d) to identify a NOAEL in Phase 2.
* Indicates instances where statistical significance ($p < 0.05$) compared to controls was reported by study authors.
[a]Excludes animals that were missing and presumed cannibalized or where vital status at birth was uncertain due to maternal cannibalization or autolysis.
[b]Excludes offspring mortalities that occurred following death of the dam; deaths assumed not treatment related.

Offspring Body Weight

Offspring body weights were available from two developmental studies (Iwai and Hoberman, 2014; Loveless et al., 2009) and a one-generation reproductive study (Loveless et al., 2009). In mice, offspring body weights were statistically significantly decreased at PND 0–7 in animals exposed gestationally (GD 6–18) to ≥100 mg/kg-day PFHxA ammonium salt. These effects were observed across two experimental cohorts with different but overlapping dose ranges (cohort 1 = 0, 100, 350, and 500 mg/kg-day; cohort 1 = 0, 7, 35, and 175 mg/kg-day). Although there is some variability in the magnitude of the dose response, consistent body weight deficits of ≥5% relative to control, a level of change that may be biologically significant during early development (U.S. EPA, 2012a, 1991), generally persisted to the end of lactation (Table 3-14). Some of this variability may be explained by differences in control body weights across the two cohorts or higher mortality of low body weight pups in the high dose groups. After weaning, some body weight deficits persisted, with females with the 350 mg/kg-day dose group showing a statistically significant reduction through the end of the experiment (PND 41) (Iwai and Hoberman, 2014).

Similar results were reported in two experiments with rats exposed to PFHxA sodium salt (Loveless et al., 2009). In the developmental study, fetal body weights (GD 21) of animals exposed gestationally (GD 6–20) to 500 mg/kg-day were decreased by 9% relative to controls, although this

change was not statistically significant, but no effects were observed at the lower doses. In the one-generation reproductive study, a dose-related decrease (4%, 11%, and 18% decrease relative to controls for 20, 100, and 500 mg/kg-day, respectively, reaching statistical significance at the highest dose) was found in pup body weights across all dose groups at PND 0. Similar to results in the mouse study (Iwai and Hoberman, 2014), body weight deficits ≥5% relative to control were observed through the end of lactation (PND 21) in the 100 and 500 mg/kg-d dose groups, but resolved after weaning (Loveless et al., 2009).

Neither study reported treatment-related effects on body weight change (i.e., gains or losses) between weaning and the end of testing (PND 21–41 for mice; PND 21–60 for rats) (Iwai and Hoberman, 2014; Loveless et al., 2009). Results are presented in Figure 3-8 and Table 3-14.

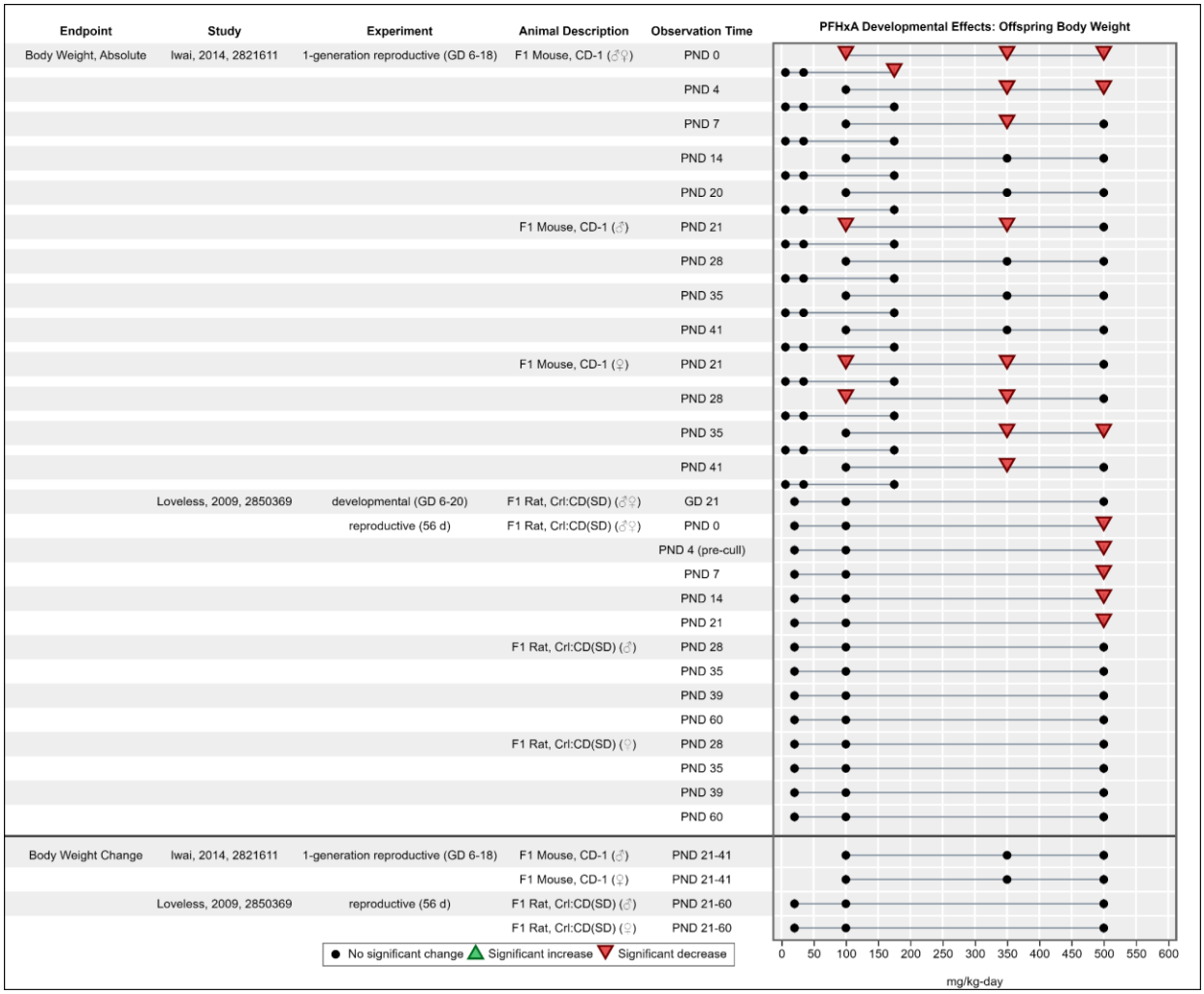


Figure 3-8. Developmental effects on offspring body weight in mice exposed to PFHxA ammonium salt and rats exposed to PFHxA sodium salt (HAWC: PFHxA – Animal Toxicity Developmental Effects link).

Table 3-14. Percent change relative to control in offspring body weight due to PFHxA sodium or ammonium salt exposure in developmental oral toxicity studies

Postnatal date (GD 6–18) and sex (Iwai and Hoberman, 2014)	Dose (mg/kg-d)						
	7	20	35	100	175	350	500
PND 0, male and female (combined) mice	0		0	−6*	−13*	−13*	−13*
PND 4, male and female (combined) mice	0		7	−7	−4*	−27*	−20*
PND 7, male and female (combined) mice	0		5	−7	0	−18*	−11
PND 14, male and female (combined) mice	−1		3	−8	0	−14	−8
PND 20, male and female (combined) mice	−2		6	−11	2	−20	−12
PND 21, male mice	3		4	−15*	−1	−18*	−14
PND 28, male mice	2		3	−10	0	−10	−8
PND 35, male mice	1		1	−4	−1	−3	−5
PND 41, male mice	1		−1	−2	−3	−3	−4
PND 21, female mice	0		6	−14*	1	−17*	−8
PND 28, female mice	0		4	−9*	−1	−16*	−7
PND 35, female mice	−1		2	−4	−1	−10*	−7*
PND 41, female mice	−3		−1	−4	−3	−8*	−4
Fetal body weight, developmental exposure (GD 6–20) (Loveless et al., 2009)							
GD 21, male and female (combined) rats		−2		0			−9
Postnatal body weight, one-generation reproductive exposure (Loveless et al., 2009)							
PND 0, male and female (combined) rats		−4		−11			−18*
PND 7, male and female (combined) rats		0		−6			−17*
PND 14, male and female (combined) rats		3		−6			−17*
PND 21, male and female (combined) rats		3		−5			−18*
PND 28, male rats		2		−1			−5
PND 35, male rats		1		−1			−3
PND 39, male rats		2		−1			−3
PND 60, male rats		2		−1			−3
PND 28, female rats		1		−5			−4
PND 35, female rats		1		−4			−1
PND 39, female rats		−1		−5			−3
PND 60, female rats		−1		−5			−3

* Indicates instances where statistical significance ($p < 0.05$) compared to controls was reported by study authors; shaded cells represent doses not included in the individual studies.

Eye Opening

Potential effects of PFHxA exposure on eye opening were evaluated in a developmental study in mice (Iwai and Hoberman, 2014). On PND 14, Iwai and Hoberman (2014) reported a statistically significant delay in eye opening, with less than 50% of pups in the 350 and 500 mg/kg-day PFHxA ammonium salt exposure groups having reached this milestone compared to 85% among vehicle controls (see Figure 3-9). Although pup body weight changes were not statistically significantly at this timepoint, they were decrements of a magnitude considered to be potentially biologically significant (8%–14%) and some developmental landmarks are correlated with postnatal body weight gain (U.S. EPA, 2016c). Delays in eye opening persisted in the 350 and 500 mg/kg-day dose groups on PND 15 but were not statistically significant. Eye opening in mice typically occurs between PND 11 and PND 14, with full functionality a few days later (Brust et al., 2015). Delays in eye opening can have long-term impacts on vision by interfering with sensory input during the critical window of visual cortex development (Espinosa and Stryker, 2012; Wiesel, 1982). The results are summarized in Figure 3-9 and Table 3-15.

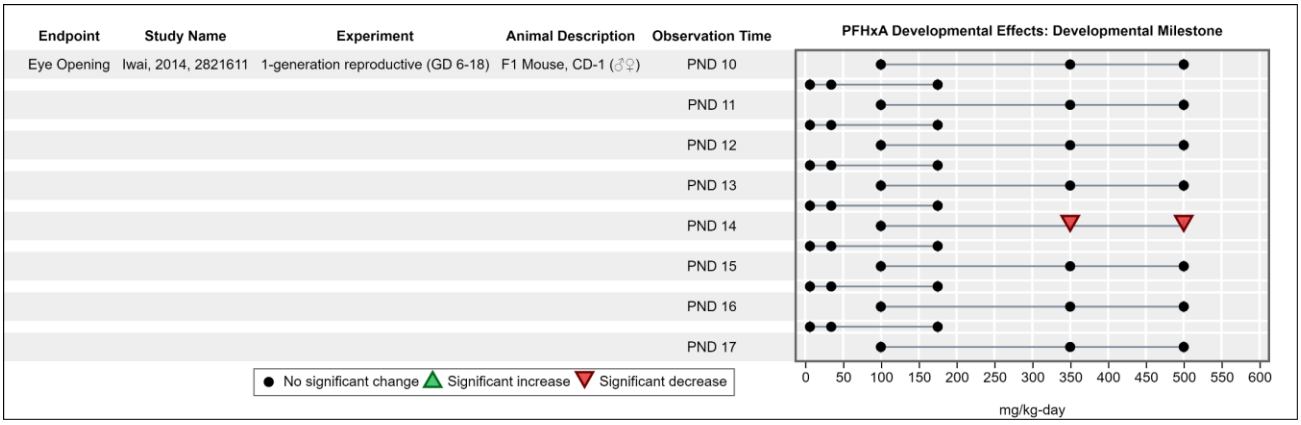


Figure 3-9. Developmental effects on eye opening (percent change relative to control) in mice exposed to PFHxA ammonium salt (HAWC: PFHxA – Animal Toxicity Developmental Eye Effects link).

Table 3-15. Percent change relative to control in eye opening due to PFHxA ammonium salt exposure in a developmental oral toxicity study

Study design and reference	Dose (mg/kg-d)					
	7	35	100	175	350	500
PND 13, male and female (combined) mice (Iwai and Hoberman, 2014)	−6	34	−56	−21	−58	−55
PND 14, male and female (combined) mice (Iwai and Hoberman, 2014)	2	4	−17	−8	−49*	−39*
PND 15, male and female (combined) mice (Iwai and Hoberman, 2014)	0	0	−10	−5	−23	−25
PND 16, male and female (combined) mice (Iwai and Hoberman, 2014)	0	0	−1	0	−9	−1

* Indicates instances where statistical significance ($p < 0.05$) compared to controls was reported by study authors.

Malformations and Variations

Potential effects of PFHxA exposure on fetal malformations and variations were evaluated in a single developmental study (Loveless et al., 2009). No treatment-related effects were found on fetal malformations or variations in rats following gestational (GD 6–20) exposure to up to 500 mg/kg-day PFHxA sodium salt.

Evidence Integration

The only available human study examining potential developmental effects was considered *uninformative*; therefore, there is *indeterminate* human evidence of developmental effects.

In animals, three *high* confidence studies reported in two publications examined developmental effects following maternal perinatal exposure to PFHxA salts (Iwai and Hoberman, 2014; Loveless et al., 2009). Treatment-related effects, including decreased offspring body weight, increased mortality, and delayed eye opening, were observed in mice following exposure to PFHxA ammonium salt at doses as low as 100 mg/kg-day (Iwai and Hoberman, 2014). Reductions in offspring body weight were also found in the one-generation reproductive and developmental studies in rats, although effects were less pronounced than those observed in mice. Animals in the reproductive cohort exposed throughout gestation and lactation showed body weight reductions that may be biologically significant (≥5%) at exposure to ≥100 mg/kg-day and statistically significant at 500 mg/kg-day that persisted to PND 21, whereas the developmental cohort was reduced (9%) only at the high dose (500 mg/kg-day).

In general, effects on development were greatest in magnitude from PND 0 to PND 7, suggesting that the early postnatal period might be a sensitive window for developmental effects associated with PFHxA exposure. Although the evidence base is small, the data are strengthened by coherent evidence across outcomes, consistency of effects on body weight across two species/studies, and the severity of some of the outcomes (e.g., increased offspring mortality). In addition, a similar pattern of effects on development (i.e., offspring body weight reductions and delays in developmental milestones) has been observed with other PFAS, including PFBS (U.S. EPA, 2021b) and PFBA (U.S. EPA, 2022a), providing additional support for these specific findings. Importantly, reduced growth during early life is associated with increased risk of developing adverse health effects in humans, including cardiovascular disease, type 2 diabetes, and early mortality in later life (Thompson and Regnault, 2011; Kajantie et al., 2005; Barker, 2004). Similarly, delays in eye opening can have lasting adverse impacts preventing visual input during a critical period of development for the visual cortex (Espinosa and Stryker, 2012; Wiesel, 1982).

The potential for maternal effects to act as a driver for the observed developmental effects was considered. In Iwai and Hoberman (2014), there was no effect on maternal body weight in either cohort. Some deaths were observed among the cohort 1 dams, but these did not appear to be dose related as there was aa similar incidence observed across controls and exposed groups. Reductions in maternal body weight were noted in the developmental study by Loveless et al.

(2009). Dams exposed to 500 mg/kg-day from GD 6–20 showed a slight but statistically significant 5% decrease in total net body weight (i.e., terminal body weight minus the gravid uterine weight) and body weight gain on GD 21 (Loveless et al., 2009). In the one-generation reproductive study, Loveless et al. (2009) reported a statistically significant reduction in maternal weight *gain* in the highest dose group (500 mg/kg-day); however, this effect was limited to early gestation (GD 0–7). Importantly, there was no effect on maternal body weight gain over the entire gestational window (GD 0–21), nor was there any observed effects on total or net maternal body weights. Thus, the effects on offspring body weight in this study are not expected to be driven by maternal toxicity. Given this interpretation of an effect on development and based on the multiple adverse changes in pups, there is *moderate* animal evidence of developmental effects.

Overall, the currently available ***evidence indicates*** that PFHxA likely causes developmental effects in humans given sufficient exposure conditions (see Table 3-16).[6] This judgment is based primarily on several *high* confidence, gestational exposure experiments in mice (and supportive findings in rats), with effects occurring after oral PFHxA exposure at ≥100 mg/kg-day. These findings are interpreted as relevant to humans in the absence of evidence to the contrary. This assumption is based on Guidelines for Developmental Toxicity Risk Assessment (U.S. EPA, 1991).

[6] The "sufficient exposure conditions" are more fully evaluated and defined for the identified health effects through dose-response analysis in Section 5.

Table 3-16. Evidence profile table for developmental effects

<table>
<tr><th colspan="5">Evidence stream summary and interpretation</th><th>Evidence integration summary judgment</th></tr>
<tr><td colspan="5">Evidence from studies of exposed humans</td><td rowspan="11">⊕⊕⊙
Evidence indicates (likely)

Primary basis:
Three high confidence studies in rats and mice including gestational (rats and mice) and continuous one-generational reproductive (rats) exposures, generally observing effects at ≥100 mg/kg-d PFHxA ammonium or sodium salt.

Human relevance:
Without evidence to the contrary, effects in rats and mice are considered relevant to humans.

Cross stream coherence:
N/A (human evidence indeterminate).

• Susceptible populations and lifestages:</td></tr>
<tr><td>Studies and confidence</td><td>Factors that increase certainty</td><td>Factors that decrease certainty</td><td>Summary and key findings</td><td>Evidence stream judgment</td></tr>
<tr><td colspan="4">• There were no informative human studies available from the PFHxA evidence base.</td><td>⊙⊙⊙
Indeterminate</td></tr>
<tr><td colspan="5">Evidence from animal studies</td></tr>
<tr><td>Studies and confidence</td><td>Factors that increase certainty</td><td>Factors that decrease certainty</td><td>Summary and key findings</td><td>Evidence stream judgment</td></tr>
<tr><td>Offspring Mortality
2 high confidence studies in rats and mice:
• GD 6–18 (mice)
• 1-generation reproductive (rats)</td><td>• High confidence studies
• Concerning severity of effect – increased mortality</td><td>• Unexplained inconsistency across species</td><td>• Increased perinatal mortality at ≥350 mg/kg-d in mice</td><td rowspan="3">⊕⊕⊙
Moderate
Developmental effects observed in multiple high confidence studies conducted in two species exposed to different PFHxA salts under different exposure scenarios. Effects on body weight were observed at doses that were not associated with offspring mortality or maternal toxicity.</td></tr>
<tr><td>Body Weight
3 high confidence studies in rats and mice:
• GD 6–18 (mice)
• GD 6–20 (rats)
• 1-generation reproductive (rats)</td><td>• High confidence studies
• Consistency across studies and species
• Dose response observed in mouse study</td><td>• No factors noted</td><td>• Postnatal body weight decreased at ≥100 mg/kg-d in rats and mice
• Fetal body weight decreased at 500 mg/kg-d in rats</td></tr>
<tr><td>Eye Opening
1 high confidence study in mice:</td><td>• High confidence study</td><td>• No factors noted</td><td>• Eye opening was delayed in mice prenatally exposed to</td></tr>
</table>

Evidence stream summary and interpretation					Evidence integration summary judgment
• GD 6–18			PFHxA ammonium salt at ≥350 mg/kg-d		• The available evidence indicates that development may be a susceptible lifestage for exposure to PFHxA.
Malformations and variations 1 high confidence study in rats: • GD 6–20	• *High* confidence study.	• No factors noted.	• No fetal malformations or variations observed at ≤500 mg/kg-d		
Mechanistic evidence and supplemental information					
Biological events or pathways	**Summary of key findings, limitations, and interpretation**			**Evidence stream summary**	
• There were no informative mechanistic studies available from the PFHxA evidence base.					

3.2.3. Renal Effects

Human

Three epidemiological studies investigated the relationship between PFHxA exposure and effects on the renal system. Two cross-sectional studies of adults in Korea and older adults in China (Zhang et al., 2019; Seo et al., 2018) were considered *uninformative* due to lack of consideration of confounding, including age and sex. The remaining study was a cross-sectional study of primarily government employees in China (Wang et al., 2019) and was classified as *low* confidence primarily due to significant concerns for reverse causality that could result if there is decreased elimination of PFAS with reduced renal function (this is also a concern for the *uninformative* studies). They observed a significant decrease in estimated glomerular filtration rate (eGFR) with higher serum PFHxA levels (β: −0.3 change in eGFR as mL/min/1.73 m^2 per 1 ln-unit PFHxA [95% CI: −0.6, −0.01]). No association was observed with chronic kidney disease. Due to the potential for reverse causality and limited sensitivity due to narrow exposure contrast for PFHxA, there is substantial uncertainty in the results of this study. A summary of the study evaluations is presented in Figure 3-10, and additional details can be obtained by clicking the HAWC link.

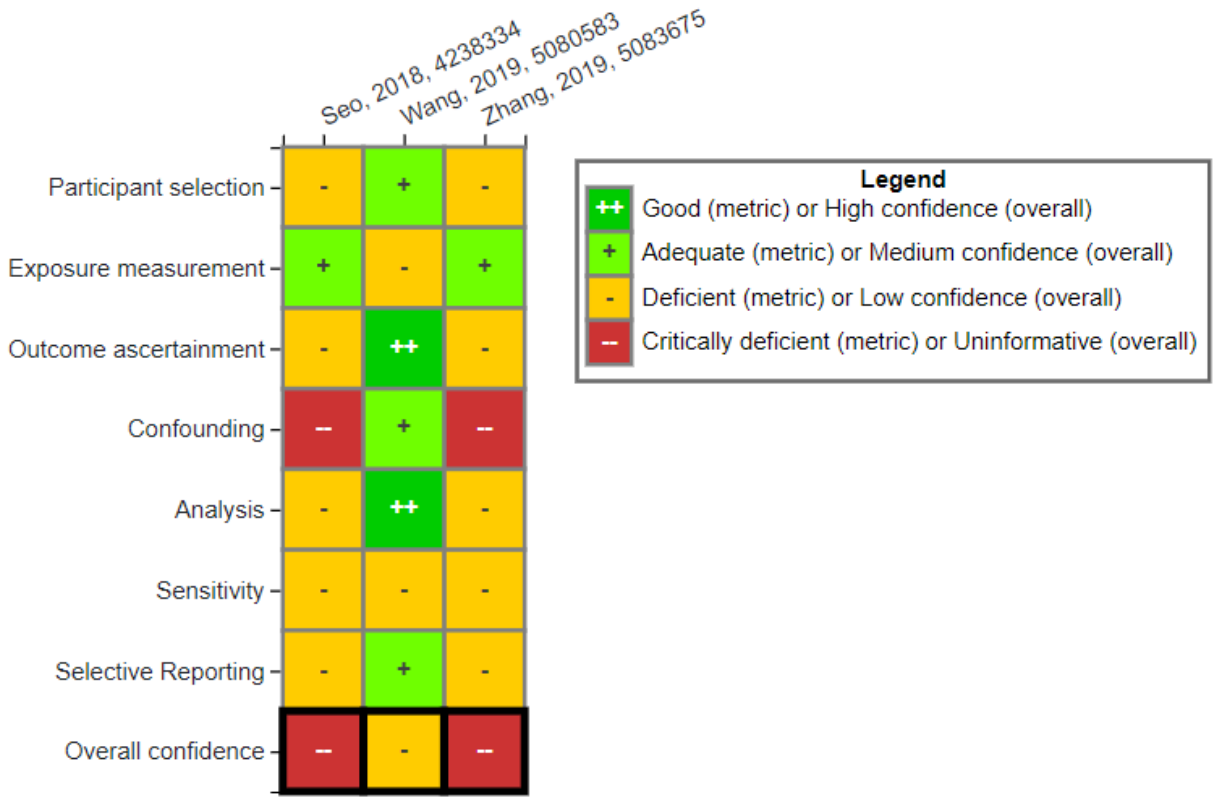


Figure 3-10. Study evaluation for human epidemiological studies reporting findings from PFHxA exposures (full details available by clicking HAWC link).

Animal

Four short-term (28-day), subchronic, or chronic animal studies evaluated potential renal effects of PFHxA or PFHxA sodium salt in rats. Most of the outcome-specific study ratings were rated *high* confidence. For Chengelis et al. (2009b), limitations were identified that influenced some outcome-specific ratings. Specifically, histopathology was rated *low* confidence because of issues related to observational bias, endpoint sensitivity and specificity, and results presentation. Urinary biomarker outcomes in the same study were rated *medium* confidence because of results presentation (only qualitative results were reported). The results of the outcome-specific confidence judgments are summarized in Table 3-17, and full study evaluation details can be viewed by clicking the HAWC link.

Table 3-17. Renal endpoints for PFHxA and associated confidence scores from repeated-dose animal toxicity studies

Author (year)	Species, strain (sex)	Exposure design	Exposure route	Blood biomarkers	Urinary biomarkers	Histopathology	Organ weight
NTP (2018)	Rat, Harlan Sprague-Dawley (male and female)	Short term (28 d)	Gavage[a] Male and female: 0, 62.5, 125, 250, 500, 1,000 mg/kg-d	++	NM	++	++
Chengelis et al. (2009b)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	Subchronic (90 d)	Gavage[a] Male and female: 0, 10, 50, 200 mg/kg-d	NR	+	–	++
Loveless et al. (2009)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	Subchronic (90 d)	Gavage[b] Male and female: 0, 20, 100, 500 mg/kg-d	++	++	++	++
Klaunig et al. (2015)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	2-yr cancer bioassay	Gavage[a] Male: 0, 2.5, 15, 100 mg/kg-d Female: 0, 5, 30, 200 mg/kg-d	++	++	++	NM

++ Outcome rating of *high* confidence; + outcome rating of *medium* confidence; – outcome rating of *low* confidence; NR, outcome not reported; NM, outcome not measured.

[a,b]Study evaluation for animal toxicological renal endpoints reported from studies with male and female rats receiving PFHxA[a] or PFHxA sodium salt[b] by gavage. Study evaluation details for all outcomes are available by clicking the HAWC link.

Organ Weight

Increases in relative kidney weight were observed in both sexes in all three studies that reported this endpoint (NTP, 2018; Chengelis et al., 2009b; Loveless et al., 2009). There were statistically significant findings in male rat dose groups at PFHxA doses as low as 10 mg/kg-day in the subchronic study (Chengelis et al., 2009b). Except for the results from Chengelis et al. (2009b), effects on relative kidney weights generally showed a weak or no dose-response gradient (see Table 3-18). Craig et al. (2015) analyzed oral chemical exposure data extracted from subchronic and chronic rat studies and found a statistically significant correlation between absolute, but not relative, kidney weight, and kidney histopathology (even at doses where terminal body weights were decreased) for most chemicals (32/35) examined indicating that absolute kidney weight is more sensitive to the effects of chemical exposure than relative kidney weight. Absolute kidney weight was increased, but only in one of the three studies reporting on this endpoint (NTP, 2018), and only in female rats at the highest dose group (1,000 mg/kg-day). The decrease in relative, but not absolute, kidney weight could be explained by body weight gain decreases in the affected dose groups: 1,000 mg/kg-day male dose group (13% decrease) (NTP, 2018), 50 and 200 mg/kg-day male dose group [8%–11% decrease with similar trends in females (Chengelis et al., 2009b)], and 500 mg/kg-day male dose group (19% decrease, no change in females) (Loveless et al., 2009). Findings and full details of PFHxA effects on kidney weights can be viewed by clicking the HAWC link.

Table 3-18. Percent increase in relative and absolute kidney weight due to PFHxA exposure in short-term, subchronic, and chronic oral toxicity studies

Endpoint and reference	Dose (mg/kg-d)									
	10	20	50	62.5	100	125	200	250	500	1,000
Relative kidney weight 28-d, female rat (NTP, 2018)				−2		0		0	3	12*
Relative kidney weight 28-d, male rat (NTP, 2018)				0		2		2	12*	19*
Relative kidney weight 90-d, female rat (Chengelis et al., 2009b)	1		12*				7			
Relative kidney weight 90-d, male rat (Chengelis et al., 2009b)	8*		7*				9*			
Relative kidney weight 90-d, female rat (Loveless et al., 2009)		−3			5				16*	
Relative kidney weight 90-d, male rat (Loveless et al., 2009)		0			11				17*	
Absolute kidney weight, right 28-d, female rat (NTP, 2018)				−1		1		1	1	9*
Absolute kidney weight, right				2		0		1	8	3

Endpoint and reference	Dose (mg/kg-d)									
	10	20	50	62.5	100	125	200	250	500	1,000
28-d, male rat (NTP, 2018)										
Absolute kidney weight 90-d, female rat (Chengelis et al., 2009b)	0		7				4			
Absolute kidney weight 90-d, male rat (Chengelis et al., 2009b)	−1		−6				2			
Absolute kidney weight 90-d, female rat (Loveless et al., 2009)		0			1				14	
Absolute kidney weight 90-d, male rat (Loveless et al., 2009)		0			8				4	

* Indicates instances where statistical significance ($p < 0.05$) compared to controls was reported by study authors; shaded cells represent doses not included in the individual studies.

Histopathology

Renal histopathological subchronic findings were qualitatively reported as null (Chengelis et al., 2009b; Loveless et al., 2009). The short-term study findings included increases in minimal chronic progressive nephropathy (CPN) that were significant (incidence 8/10) in the 1,000 mg/kg-day female dose group (see Figure 3-11) (NTP, 2018), consistent with increased absolute kidney weight. Male renal histopathological findings from the chronic study were also null, whereas findings for female rats included increased papillary necrosis (17/70 vs. 0/60 in controls) and tubular degeneration (7/70 vs. 1/60 in controls) in the highest dose group 200 mg/kg-day (Klaunig et al., 2015). Full details are available by clicking the HAWC link.

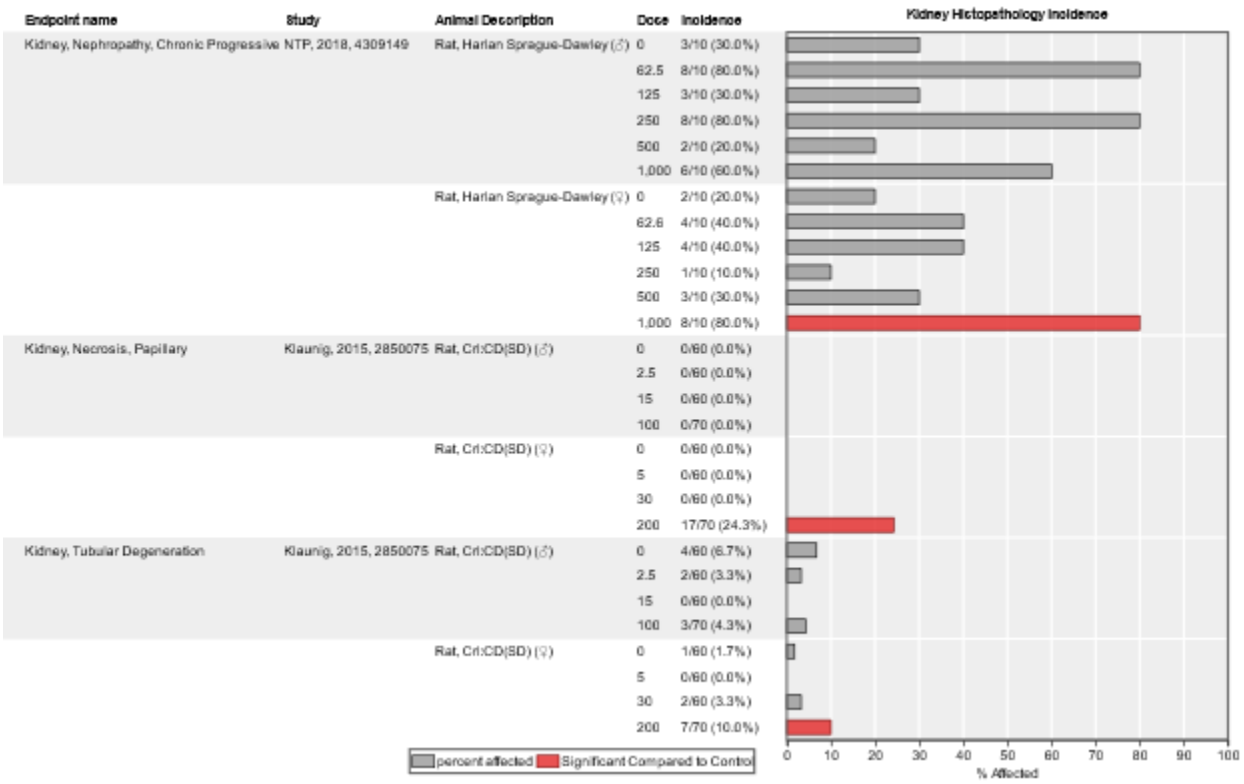


Figure 3-11. Animal toxicological renal histopathology after PFHxA exposure (full details available by clicking the HAWC link). Findings from the subchronic studies were reported as null and not included in the above visualization.

Blood and Urinary Biomarkers

Blood biomarkers of renal function were inconsistent across study designs and exposure durations. Both creatinine and blood urea nitrogen (BUN) are removed from the blood by the kidneys and often used as indicators of kidney function. Creatinine is a waste product of creatine metabolism (primarily in muscle), and BUN is a waste product of protein metabolism in the liver. No observations of changes in urea nitrogen or creatinine were reported from Chengelis et al. (2009b) and Klaunig et al. (2015). In the short-term study (NTP, 2018), BUN was unchanged in both sexes in all dose groups. Changes in creatinine were inconsistent across sexes with null findings in females, whereas a 13% decrease ($p < 0.05$) was found in the male 500 mg/kg-day dose group (NTP, 2018). In a subchronic study, Loveless et al. (2009) reported a sex-specific increase in BUN in the male 200 mg/kg-day dose group, whereas creatinine was decreased in both male and female rats dosed with 200 mg/kg-day PFHxA sodium.

Urinalysis findings included total urine volume and other measures of urine concentrating ability (e.g., specific gravity, urobiloginen). The urinalysis findings were more consistent than the blood biomarkers, but still difficult to interpret as adverse or nonadverse. Urinalysis findings were not measured in the short-term study (NTP, 2018) and were reported as null in a subchronic study (Chengelis et al., 2009b). Findings from the other subchronic study (Loveless et al., 2009) identified

changes in urine concentration reflected as decreased (50%–88%) urine protein combined with increased (207%–300%) total urine volume in males and females in the 500 mg/kg-day dose groups. Coherent with increased urine volume, osmolality was decreased (47%, $p < 0.05$), but only in the male 500 mg/kg-day dose group (Loveless et al., 2009). Urobilinogen and pH findings were null in both male and females in the subchronic study (Loveless et al., 2009). Findings from the chronic study lacked consistency between sexes and did not exhibit a clear dose-response relationship (Klaunig et al., 2015). Specifically, total urine volume was increased in the female 200 mg/kg-day dose group and null in all male dose groups. Urine specific gravity was increased ($p < 0.05$) in the male 15 mg/kg-day dose group and similar to controls in the 100 mg/kg-day dose group, although specific gravity was increased ($p < 0.05$) in the female 200 mg/kg-day dose group. Urine pH was low in males (compared to controls) only in the 100 mg/kg-day dose groups at 26 and 52 weeks (Klaunig et al., 2015). Total urine volume findings were null in males, whereas an increase was found in female rats from the 200 mg/kg-day dose group at 26 weeks that returned to control levels at 52 weeks study duration (Klaunig et al., 2015). Findings are summarized in Figure 3-12, and full details are available in the HAWC link.

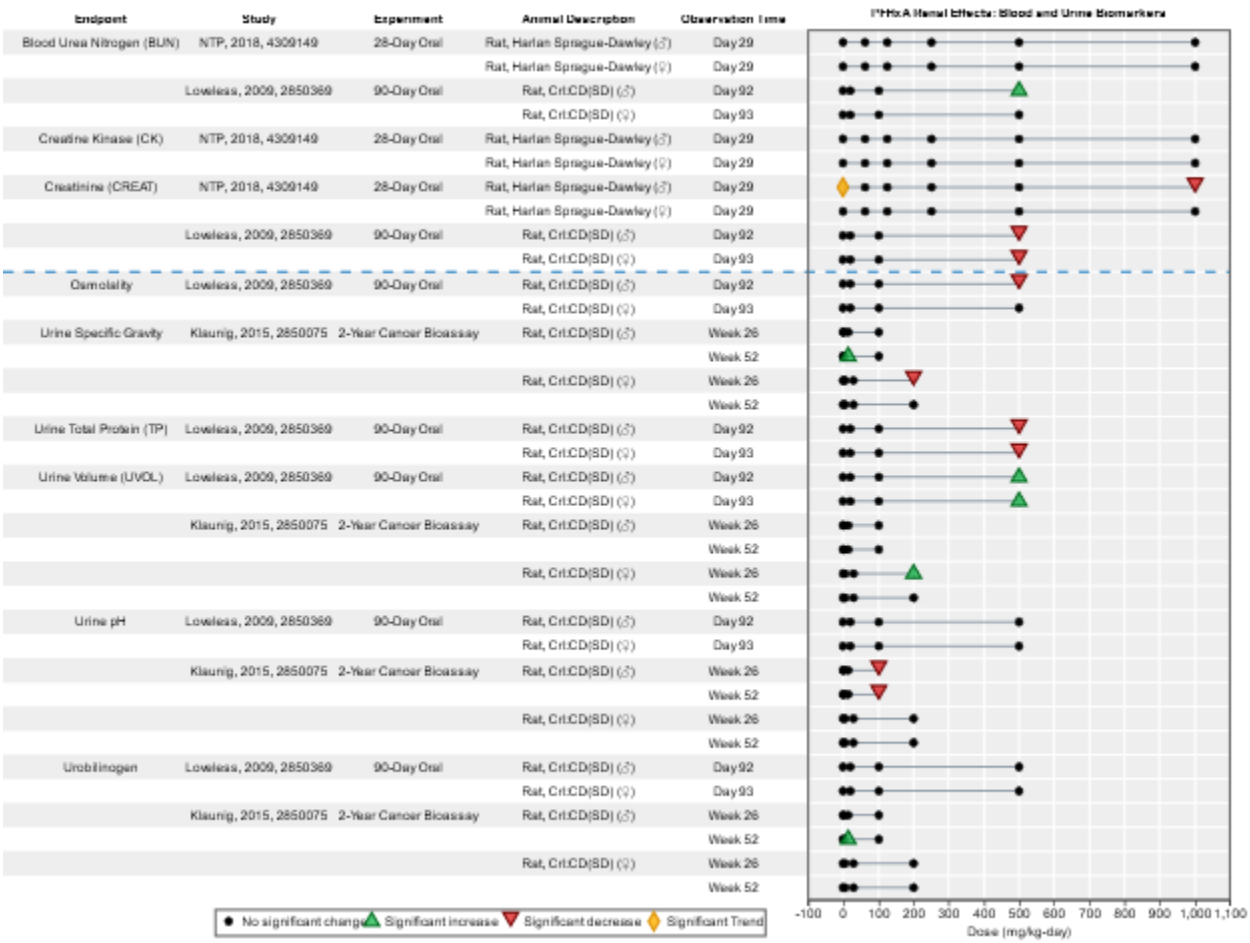


Figure 3-12. PFHxA Effects on blood and urine biomarkers of renal function (full details available by clicking the HAWC link). The dashed blue line divides blood (top) from urinary biomarkers. Note that blood urea nitrogen and creatinine were described as null, but findings were not quantitatively reported.

Evidence Integration

The human evidence was limited to a single *low* confidence study reporting an inverse association between PFHxA exposure and eGFR, although notable uncertainty in this result exists due to potential for reverse causality. Based on these data, there is *indeterminate* human evidence for renal effects.

The evidence base for renal effects in experimental animals was drawn from generally *high* confidence studies including one short-term, two subchronic studies, and one chronic study. Findings were, in general, null except for histopathology and some urinary biomarkers. Kidney histopathology was the most significant finding in the short term and chronic studies. In the short-term study, increased incidence of CPN was observed in female rats at the highest dose (1,000 mg/kg-day PFHxA) and consistent with increased absolute kidney weight (females only). Histopathological findings were null in both subchronic studies at doses up to 500 mg/kg-day. In the chronic study, the incidence of papillary necrosis and tubular degeneration were increased in

females compared to controls at the highest dose (200 mg/kg-day) that is twice the highest dose received by male rats in the chronic study. Some changes occurred in urinary biomarkers (decreased urine pH, increased urine volume) and potentially correlated changes were observed in female histopathology in the chronic study. However, inconsistencies across studies at similar observation times and doses were notable. Based on these results, there is *slight* animal evidence of renal effects.

Overall, the currently available ***evidence is inadequate*** to assess whether PFHxA may causes renal effects in humans (see Table 3-19).

Table 3-19. Evidence profile table for renal effects

Evidence stream summary and interpretation					Evidence integration summary judgment
Evidence from studies of exposed humans					⊙⊙⊙ ***Evidence inadequate*** *Primary basis:* Indeterminate evidence in humans and animal evidence is largely null or lacking biological coherence, overall effects not clearly adverse *Human relevance:* Without evidence to the contrary, effects in rats are considered relevant to humans *Cross-stream coherence:* N/A (human evidence *indeterminate*) *Susceptible lifestages:* • No evidence to inform
Studies and confidence	**Factors that increase certainty**	**Factors that decrease certainty**	**Summary and key findings**	**Evidence stream judgment**	
Kidney Function 1 *low* confidence study	• No factors noted	• Low sensitivity • Potential for reverse causality	• Weak association of PFHxA with decrease in estimated eGFR	⊙⊙⊙ ***Indeterminate***	
Evidence from animal studies					
Studies and confidence	**Factors that increase certainty**	**Factors that decrease certainty**	**Summary and key findings**	**Evidence stream judgment**	
Organ Weight 3 *high* confidence studies in adult rats: • 28-d • 90-d (2 studies)	• *Consistent* increases, all studies	• Weak Weak/no dose response	• Increased relative kidney weight at ≥10 mg/kg-d. • Increase absolute kidney weight at 1,000 mg/kg-d; 28-d study, females only	⊕⊙⊙ ***Slight*** Findings of adversity were considered uncertain based on lack of coherence between effects (organ weight, histopathology, blood, and urine biomarkers) inconsistency between sexes, and lack of coherence across exposure designs	
Histopathology 3 *high* confidence studies in adult rats: • 28-d • 90-d • 2-yr 1 *low* confidence study in adult rats: • 90-d	• *Large magnitude of effect*, up to 24.3% for papillary necrosis; up to 80% for chronic progressive nephropathy	• No factors noted	• Increased incidence papillary necrosis, tubular degeneration, chronic progressive nephropathy at ≥200 mg/kg-d; female rats only, 28-d and chronic studies		

Evidence stream summary and interpretation					Evidence integration summary judgment
Blood Biomarkers 4 *high* confidence studies in adult rats: • 28-d • 90-d (2 studies) • 2-yr	• No factors noted	• *Lack of coherence* with other histopathological findings; chronic study • Lack of consistent effect across studies	• Increased BUN at 500 mg/kg-d; males only, 90-d study. • Decreased creatinine at ≥500 mg/kg-d), both sexes, 1 subchronic study • Decreased creatine at 1,000 mg/kg-d; males only, 28-d study • No treatment related creatinine kinase findings; both sexes, 28-d study		
Urinary Biomarkers 3 *high* confidence studies in adult rats: • 28-d • 90-d • 2-yr 1 *medium* confidence study in adult rats: • 90-d	• *Coherence* of urine protein, urine volume, urine specific gravity, and decreased osmolality	• *Lack of coherence* with histopathological findings.	• Decreased osmolality 500 mg/kg-d; males only, 1 subchronic study • Decreased urine protein and increased urine volume in at 500 kg/kg-d; both sexes, 1 subchronic study • Increased total urine volume at ≥200 mg/kg-d; both sexes — 1 subchronic study, females only, 1 2-yr study • Decreased urine pH at 100 mg/kg-d; males only, 1 2-yr study		

Evidence stream summary and interpretation					Evidence integration summary judgment
			• No treatment related findings for urobilinogen; both sexes, 1 subchronic study and 1 2-yr study		
Mechanistic evidence and supplemental information					
Biological events or pathways	**Primary evidence evaluated Key findings, interpretation, and limitations**			**Evidence stream summary**	
Molecular Events—Oatp1a1	*Key findings and interpretation:* Sex hormone-dependent regulation of Oatp1a1 mRNA and protein level (see Section 3.1.4).			Sex-specific Oatp1a1 expression appears to lead to sex-specific PFHxA elimination, leading to longer PFHxA half-life in male rats compared with females, which may explain sex-specific renal findings.	

3.2.4. Hematopoietic Effects

Hematology is a subgroup of clinical pathological parameters concerned with morphology, physiology, and pathology of blood and blood-forming tissues. Hematological parameters are measured using blood tests such as complete blood counts (CBC) and a clinical chemistry panel. The CBC is a blood test that measures (red blood cells, white blood cells, hemoglobin, hematocrit, and platelets), whereas the clinical chemistry panel measures the proteins, enzymes, chemicals, and waste products in the blood. Hematological measures, when evaluated together and with information on other biomarkers, are informative diagnostic tests for blood-forming tissues (i.e., bone marrow, spleen, liver) and organ function. In animals, blood tests are influenced by the feeding protocol, blood collection site, animal age, and other factors.

Human Studies

One human study (Jiang et al., 2014) evaluated blood counts in samples drawn from a population of 141 pregnant women living in Tianjin, China. The study was considered *uninformative*, however, due to lack of consideration of confounding, including age, which is expected to substantially bias the results. Full study evaluation for Jiang et al. (2014) is available by clicking the HAWC link.

Animal Studies

Several animal toxicological studies were available that assessed hematopoietic parameters including a *high* confidence short-term study (NTP, 2018), *high* confidence (Chengelis et al., 2009b) and *high* confidence (Loveless et al., 2009) subchronic studies, and a *high* confidence chronic study (Klaunig et al., 2015). Cell counts for the blood components associated with immune system responses are summarized under in Immune Effects, see Section 3.2.8. Study findings are discussed below and summarized in Table 3-20 (full details are available by clicking the HAWC link), and summary details are available in PFHxA Tableau visualization.

Table 3-20. Hematopoietic endpoints for PFHxA and associated confidence scores from repeated-dose animal toxicity studies

Author (year)	Species, strain (sex)	Exposure design	Exposure route and dose range	Hematology and hemostasis
NTP (2018)	Rat, Harlan Sprague-Dawley (male and female)	Short term (28 d)	Gavage[a] Male and female: 0, 62.5, 125, 250, 500, 1,000 mg/kg-d	++
Chengelis et al. (2009b)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	Subchronic (90 d)	Gavage[a] Male and female: 0, 10, 50, 200 mg/kg-d	++

Author (year)	Species, strain (sex)	Exposure design	Exposure route and dose range	Hematology and hemostasis
Loveless et al. (2009)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	Subchronic (90 d)	Gavage[b] Male and female: 0, 20, 100, 500 mg/kg-d	++
Klaunig et al. (2015)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	2-yr cancer bioassay	Gavage[a] Male: 0, 2.5, 15, 100 mg/kg-d Female: 0, 5, 30, 200 mg/kg-d	++

++ Outcome rating of *high* confidence.

[a,b]Study evaluation for animal toxicological hematopoietic endpoints reported from studies with male and female rats receiving PFHxA[a] or PFHxA sodium salt[b] by gavage. Study evaluation details for all outcomes are available by clicking the HAWC link.

Hematology

Several findings were consistent (i.e., decreased red blood cells [RBCs], hematocrit, and hemoglobin) across studies and study designs that, when interpreted together, suggest PFHxA-related adverse hematological effects such as anemia (see Figure 3-13). Indications were also present that red blood cells were swollen and made up a larger proportion of the blood volume (increased mean cell volume [MCV, a measure of the average red blood cell size]). These changes were correlated with potential secondary erythrogenic responses to PFHxA exposure including increased reticulocyte (immature RBCs) counts that were consistently increased >10% across study designs and exposure durations, including the chronic study Klaunig et al. (2015) where the highest dose levels were 2–5 times lower than those tested in the subchronic studies. Specifically, a dose-responsive decrease occurred in red blood cells (see Table 3-21), hematocrit (see Table 3-22), and hemoglobin (see Table 3-23) in the short-term study with decreases at doses ranging from 62.5 mg/kg-day in male rats to 250 mg/kg-day in female rats (NTP, 2018). These findings also were observed in both subchronic studies in the highest dose groups [200 mg/kg-day in males only (Chengelis et al., 2009b) and 500 mg/kg-day in both sexes (Loveless et al., 2009)]. Of note, decreases in both hemoglobin and hematocrit were 1.5–2-fold greater in the subchronic study (Loveless et al., 2009) than in the short-term study (NTP, 2018) for both males and females at the same dose (500 mg/kg-day).

Findings from the chronic study (Klaunig et al., 2015) were generally null or observed at dose levels ≥100 mg/kg-day (100 mg/kg-day in males and 200 mg/kg-day in females) at 25 and 51 weeks. Quantitative measures of hematology beyond 52 weeks in the chronic study might be complicated due to natural diseases occurring in rodents and test variability leading to decreased sensitivity and increasing variability with the results (AACC, 1992). Klaunig et al. (2015) did, however, qualitatively evaluate blood and reported no PFHxA treatment effects on blood smear morphology. Loveless et al. (2009) also evaluated blood smears up to test day 92 with PFHxA

sodium salt exposure and noted nucleated blood cells in smears indicative of bone marrow damage or stress, but only for one female and one male.

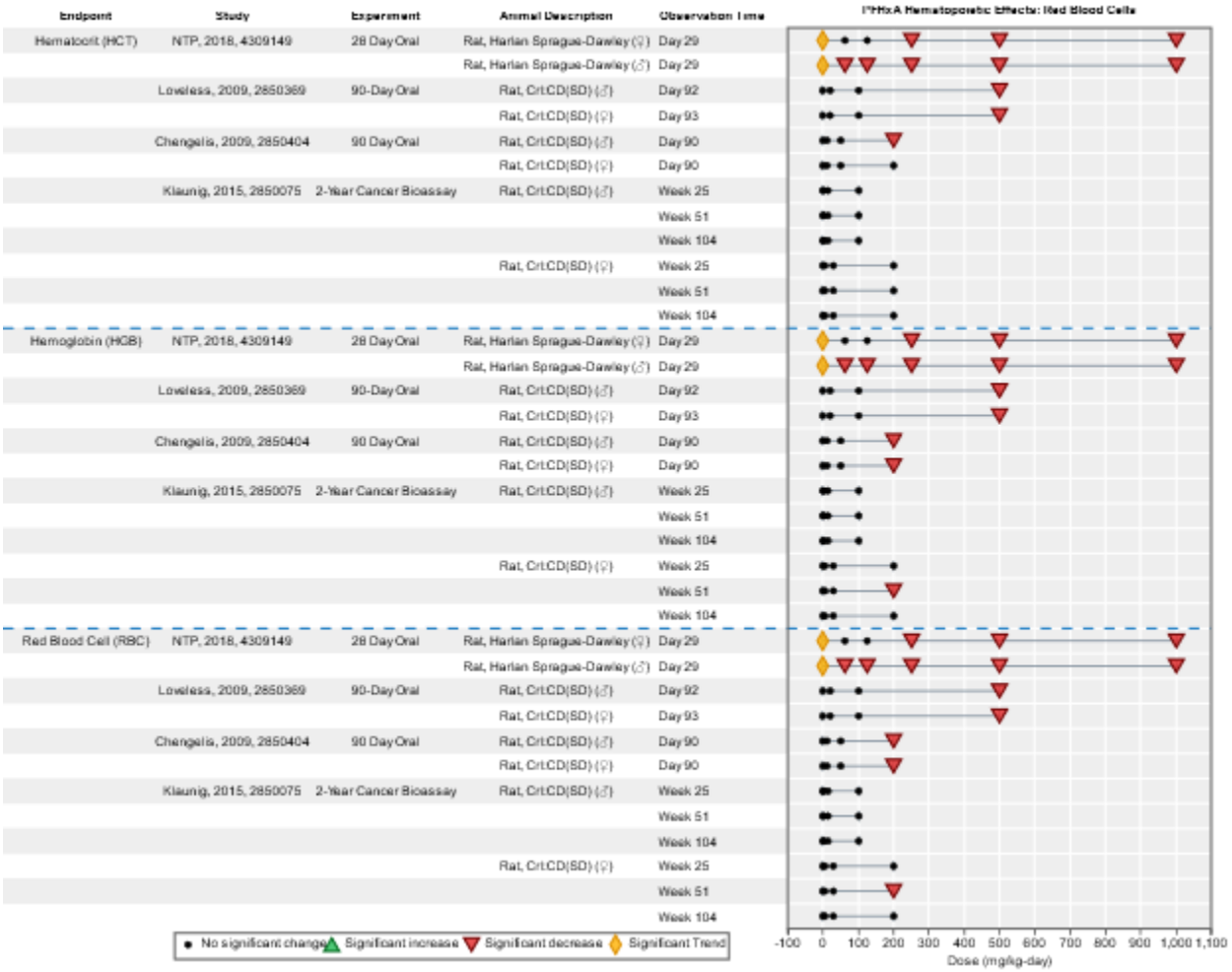


Figure 3-13. Hematological findings (HCT, HGB, and RBC) in rats exposed by gavage to PFHxA or PFHxA sodium salt (full details available by clicking the HAWC link).

HCT = hematocrit; HGB = hemoglobin; RBC = red blood cells.

Table 3-21. Percent change in red blood cells due to PFHxA exposure in short-term, subchronic, and chronic oral toxicity studies

Study design and reference	Dose (mg/kg-d)													
	2.5	5	10	15	20	30	50	62.5	100	125	200	250	500	1,000
28-d, female rat (NTP, 2018)								−1		2		−7	−10*	−26*
28-d, male rat (NTP, 2018)								−5		−5		−9*	−23*	−48*

Study design and reference	Dose (mg/kg-d)													
	2.5	5	10	15	20	30	50	62.5	100	125	200	250	500	1,000
90-d, female rat (Chengelis et al., 2009b)			−1				−3				−8*			
90-d, male rat (Chengelis et al., 2009b)			−1				0				−8*			
90-d, female rat (Loveless et al., 2009)					2				0				−18*	
90-d, male rat (Loveless et al., 2009)					1				−5				−31*	
2-yr, female rat (Klaunig et al., 2015), Wk 25		4				−2					−1			
2-yr, male rat (Klaunig et al., 2015), Wk 25	−3			−3					−4					
2-yr, female rat (Klaunig et al., 2015), Wk 51		1				0					−8*			
2-yr, male rat (Klaunig et al., 2015), Wk 51	−4			−6					−4					
2-yr, female rat (Klaunig et al., 2015), Wk 104		−1				−2					1			
2-yr, male rat (Klaunig et al., 2015), Wk 104	−7			−1					−8					

* Indicates instances where statistical significance ($p < 0.05$) compared to controls was reported by study authors; shaded cells represent doses not included in the individual studies.

The red blood cell mass parameter (MCHC, the average weight of hemoglobin in a specified volume of red blood cells) was decreased in both sexes in the short-term (NTP, 2018) and subchronic studies (Loveless et al., 2009) (see Figure 3-14). Null findings for MCHC were observed in the other subchronic study (Chengelis et al., 2009b) and the chronic study (Klaunig et al., 2015), consistent with MCHC findings at similar dose levels in the short-term (NTP, 2018) and subchronic studies (Loveless et al., 2009). MCV, a measure of average blood volume of RBCs was increased in both a short-term and a subchronic study (NTP, 2018; Loveless et al., 2009).

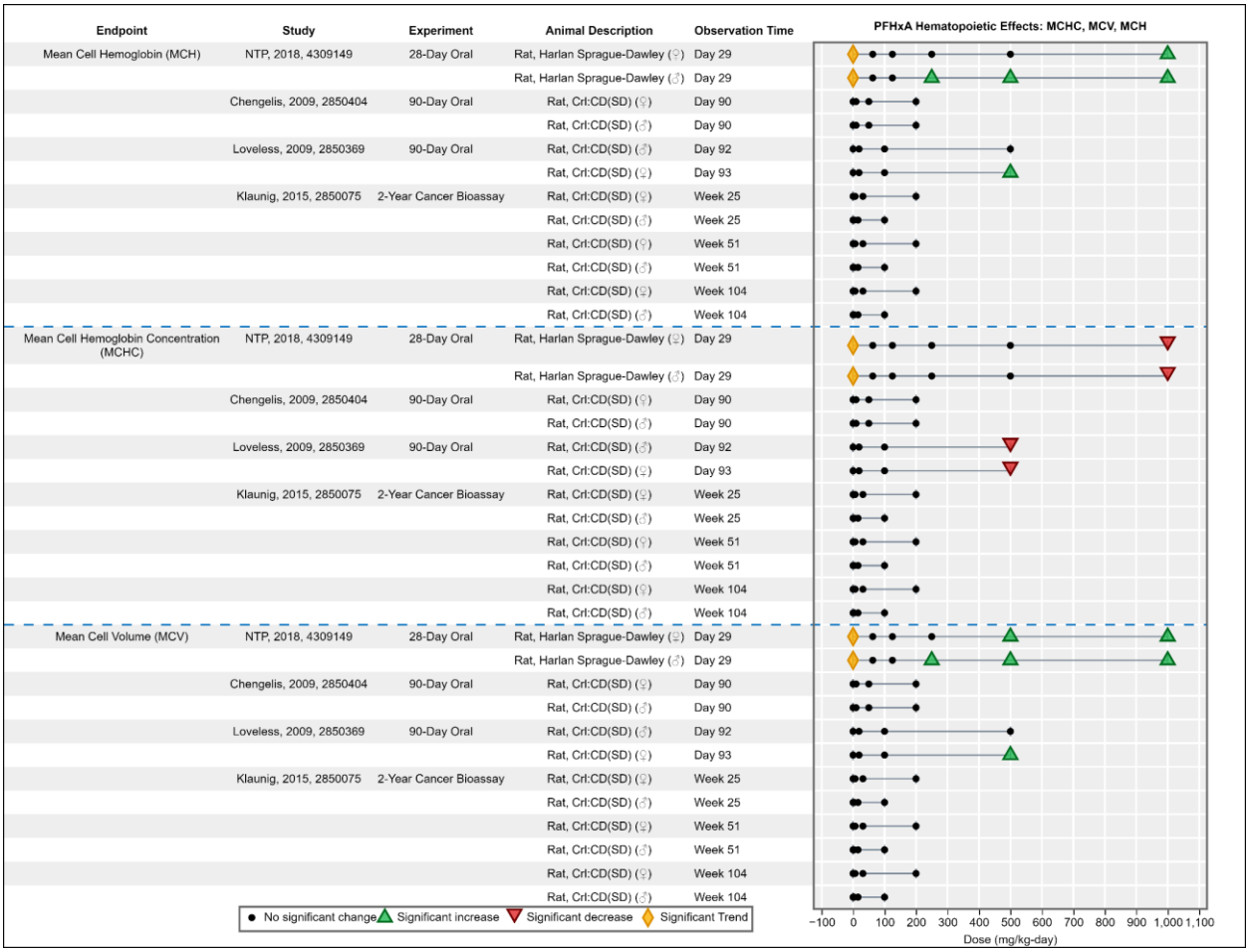


Figure 3-14. Hematological findings (MCH, MCHC, and MCV) in rats exposed by gavage to PFHxA or PFHxA sodium salt (full details available by clicking the HAWC link).

MCH = mean cell hemoglobin; MCHC = mean cell hemoglobin concentration; MCV = mean cell volume.

Table 3-22. Percent change in hematocrit due to PFHxA exposure in short-term, subchronic, and chronic oral toxicity studies

Study design and reference	Dose (mg/kg-d)													
	2.5	5	10	15	20	30	50	62.5	100	125	200	250	500	1,000
28-d, female rat (NTP, 2018)								−1		0		−7*	−8*	−17*
28-d, male rat (NTP, 2018)								−4		−6*		−6*	−17*	−30*
90-d, female rat (Chengelis et al., 2009b)			0				−5				−6			
90-d, male rat (Chengelis et al., 2009b)			−3				−3				−8			
90-d, female rat (Loveless et al., 2009)					1				0				−13*	
90-d, male rat (Loveless et al., 2009)					0				−6				−31*	

Study design and reference	Dose (mg/kg-d)													
	2.5	5	10	15	20	30	50	62.5	100	125	200	250	500	1,000
2-yr, female rat (Klaunig et al., 2015), Wk 25		3				0					0			
2-yr, male rat (Klaunig et al., 2015), Wk 25	−1			−3					−3					
2-yr, female rat (Klaunig et al., 2015), Wk 51		1				0					−4			
2-yr, male rat (Klaunig et al., 2015), Wk 51	−5			−4					−3					
2-yr, female rat (Klaunig et al., 2015), Wk 104		0				−1					1			
2-yr, male rat (Klaunig et al., 2015), Wk 104	−9			−5					−8					

* Indicates instances where statistical significance ($p < 0.05$) compared to controls was reported by study authors; shaded cells represent doses not included in the individual studies.

Table 3-23. Percent change in hemoglobin due to PFHxA exposure in short-term, subchronic, and chronic oral toxicity studies

Study design and reference	Dose (mg/kg-d)													
	2.5	5	10	15	20	30	50	62.5	100	125	200	250	500	1,000
28-d, female rat (NTP, 2018)								0		−1		−6*	−8*	−19*
28-d, male rat (NTP, 2018)								−3		−5*		−6*	−19*	−40*
90-d, female rat (Chengelis et al., 2009b)			1				−3				−6			
90-d, male rat (Chengelis et al., 2009b)			−1				−1				−8			
90-d, female rat (Loveless et al., 2009)					1				0				−15*	
90-d, male rat (Loveless et al., 2009)					1				−71				−36*	
2-yr, female rat (Klaunig et al., 2015), Wk 25		3				1					−1			
2-yr, male rat (Klaunig et al., 2015), Wk 25	−1			−2					−3					
2-yr, female rat (Klaunig et al., 2015), Wk 51		1				0					**−5***			
2-yr, male rat (Klaunig et al., 2015), Wk 51	−6			−5					−3					
2-yr, female rat (Klaunig et al., 2015), Wk 104		0				0					−1			

Study design and reference	Dose (mg/kg-d)													
	2.5	5	10	15	20	30	50	62.5	100	125	200	250	500	1,000
2-yr, male rat (Klaunig et al., 2015), Wk 104	−9			−4					−9					

* Indicates instances where statistical significance ($p < 0.05$) compared to controls was reported by study authors; shaded cells represent doses not included in the individual studies.

Increased reticulocyte (immature RBCs formed during the erythroid regenerative process) counts were consistently found across all four animal toxicological studies (see Table 3-24 and Figure 3-15) and correlated with decreases in RBCs. PFHxA treatment-related increases in reticulocyte counts were potentially a compensatory biological response to the PFHxA anemia effect. Reticulocytes were increased (>10%) across all study designs and exposure durations at 200 mg/kg-day (Klaunig et al., 2015; Chengelis et al., 2009b), 250 mg/kg-day (NTP, 2018), or 500 mg-kg/day (Loveless et al., 2009). Reticulocyte measures were also available from Klaunig et al. (2015), where increases were identified only in the high dose (200 mg/kg-day) female rat group. The observation of increased reticulocytes was coherent with histological findings of increased splenic extramedullary hematopoiesis and bone marrow erythroid hyperplasia incidence in both the males and females dosed with 500 mg/kg-day (NTP, 2018; Loveless et al., 2009) (summary details are available in PFHxA Tableau visualization). Collectively, the histological findings considered together with red blood cell parameters suggest PFHxA interacts with the erythropoietic pathways including outcomes such as anemia that can lead to compensatory erythrogenic responses in the bone marrow and spleen.

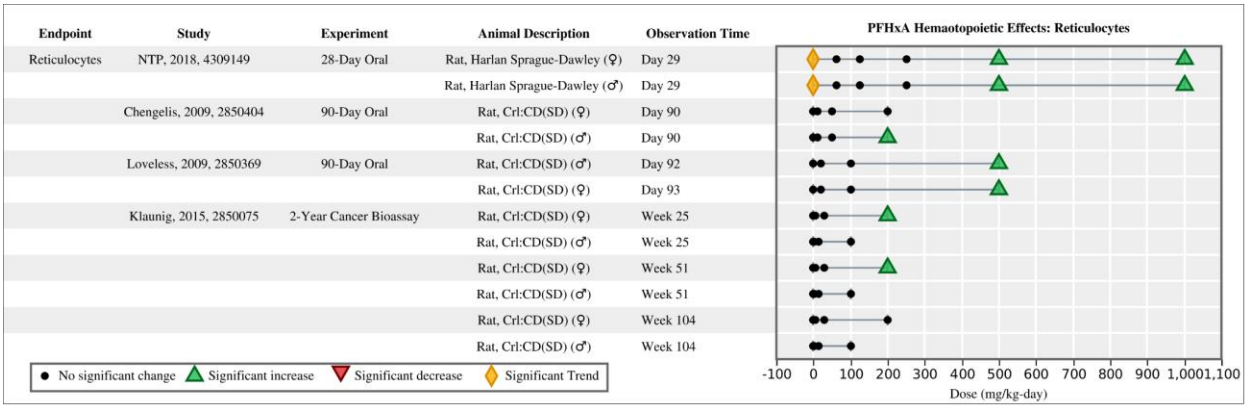


Figure 3-15. Hematological findings (reticulocytes) in rats exposed by gavage to PFHxA or PFHxA sodium salt (full details available by clicking the HAWC link).

Table 3-24. Percent change in reticulocytes due to PFHxA exposure in short-term, subchronic, and chronic oral toxicity studies

Study design and reference	Dose (mg/kg-d)													
	2.5	5	10	15	20	30	50	62.5	100	125	200	250	500	1,000
28-d, female rat (NTP, 2018)								−5		−15		15	152*	356*
28-d, male rat (NTP, 2018)								0		−2		20	89*	223*
90-d, female rat (Chengelis et al., 2009b)			−7				−13				80*			
90-d, male rat (Chengelis et al., 2009b)			−5				−13				59*			
90-d, female rat (Loveless et al., 2009)					7				13				181*	
90-d, male rat (Loveless et al., 2009)					−14				−4				210*	
2-yr, female rat (Klaunig et al., 2015), Wk 25		−5				11					26*			
2-yr, male rat (Klaunig et al., 2015), Wk 25	−5			0					15					
2-yr, female rat (Klaunig et al., 2015), Wk 51		−25				−6					56*			
2-yr, male rat (Klaunig et al., 2015), Wk 51	21			71					43					
2-yr, female rat (Klaunig et al., 2015), Wk 104		6				19					26*			
2-yr, male rat (Klaunig et al., 2015), Wk 104	21			−6					29					

* Indicates instances where statistical significance ($p < 0.05$) compared to controls was reported by study authors; shaded cells represent doses not included in the individual studies.

Hemostasis

Hemostasis findings included platelet counts, prothrombin time, and activated partial thromboplastin time. Clotting times measured by Chengelis et al. (2009b) and Klaunig et al. (2015) could be complicated because blood samples were collected from the retro-orbital sinus, a technique not recommended because it leads to prolonged clotting times that might not be reliable; thus, these endpoints were considered uninformative and are not discussed further. Findings of statistically significant increased ($p < 0.05$) platelets were observed in the short-term (NTP, 2018) and subchronic (Chengelis et al., 2009b; Loveless et al., 2009) studies in males and females dosed with 500 mg/kg-day dose (see Figure 3-16). Other hemostasis measures that included activated partial thromboplastin time (APTT) and prothrombin time (PT, a functional measure of a subset of clotting factors that contribute to APTT) were decreased inconsistently across sexes in one subchronic study (Loveless et al., 2009). PT was decreased in male dose groups receiving ≥20 mg/kg-day, whereas APTT was decreased in the 500 mg/kg-day female rat dose group. The observed increase in platelets and decreased clotting time (along with increased reticulocytes) were coherent changes indicative of an erythropoietic response in the bone marrow, suggesting bone marrow was not adversely affected by PFHxA exposure.

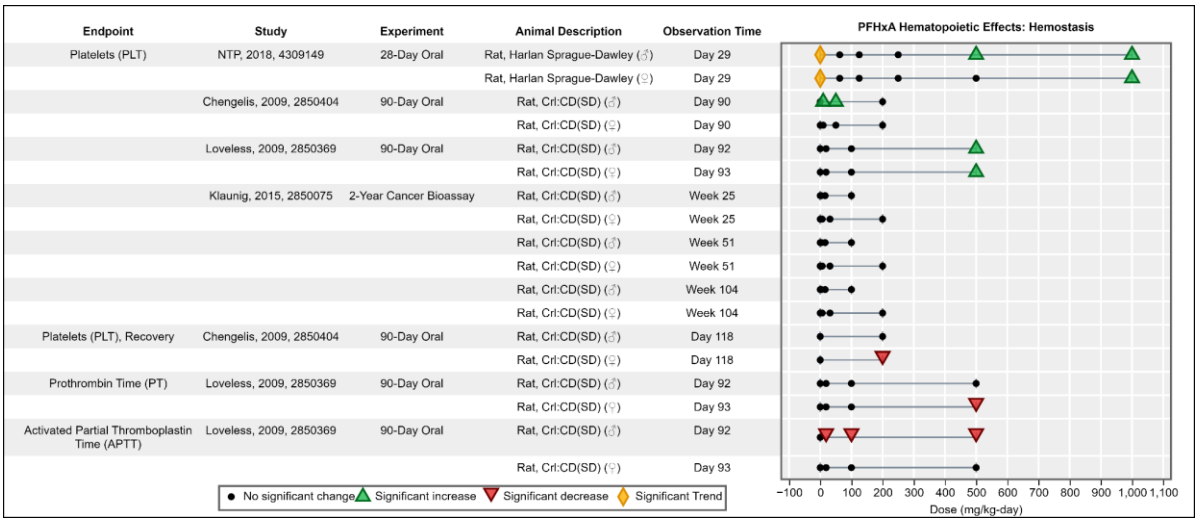


Figure 3-16. Hemostasis findings in rats exposed by gavage to PFHxA or PFHxA sodium salt (full details available by clicking the HAWC link).

Evidence Integration

The only available human study examining potential hematopoietic effects was considered *uninformative*; therefore, there is *indeterminate* human evidence of hematopoietic effects.

Collectively, the animal toxicological information provided coherent evidence indicative of macrocytic anemia (characterized by low hemoglobin and large red blood cells) that is consistent across multiple laboratories and experimental designs. Findings informing the overall judgment included consistent observations of decreased red blood cells, hematocrit, and hemoglobin at doses as low as 200 mg/kg-day generally in both sexes (summary level details are available in the Tableau link). This finding was considered an adverse response to PFHxA exposure and correlated with a compensatory increase in reticulocytes, an indicator of erythroid cell regeneration supported by histological findings of splenic extramedullary hematopoiesis and bone marrow erythroid hyperplasia. The responses across hematologic parameters in the chronic study (Klaunig et al., 2015) were only observed at the highest dose (200 mg/kg-day) in females. However, the highest dose (200 mg/kg-day in females, 100 mg/kg-day in males) in Klaunig et al. (2015) was at or lower than the observed effect level in the other available short term and subchronic studies. Further, the null responses at lower doses (2.5, 15, and 100 mg/kg-day in male rats; 5 and 30 mg/kg-day in female rats) are consistent with null responses in hematologic endpoints at similar dose levels in the short term and subchronic studies. Overall, these collective erythroid responses provide evidence for PFHxA treatment-related effects on erythropoiesis in animals.

Based on these data, there is *robust* animal evidence of hematopoietic effects. Effects on red blood cell parameters including decreased hemoglobin, decreased red blood cells, and increased reticulocytes are consistent across both subchronic and chronic studies in the 200 mg/kg-day dose groups. The animal evidence came from studies using the same outbred Crl:CD(SD) rat model originating from the same location and supplier that made available clinical hematology parameters

indicating no sex difference in the parameters reported here. Given species conservation between rodent models it was assumed that rat hematology was similar to that of the mouse where there are some hematologic differences between humans, including smaller erythrocytes, higher percentage of circulating reticulocytes (or polychromasia), physiologic splenic hematopoiesis and iron storage, and more numerous and shorter-lived erythrocytes and platelets (O'Connell et al., 2015). These differences could explain the possible regenerative response in the spleen and bone and the increase in reticulocytes (i.e., erythrogenesis and RBC turnover more rapid in rodent versus human). Therefore, while some uncertainty around the human relevance of rodent hematopoietic findings exists, these species-specific differences were controlled for in the experimental design.

Based on *indeterminate* evidence in humans and *robust* animal evidence with potentially uncertain human relevance, the currently available ***evidence indicates*** that PFHxA likely causes hematopoietic effects in humans given sufficient exposure conditions (see Table 3-25).[7] This conclusion is based on four *high* confidence studies in rats showing consistent (across durations and study types) and coherent effects (across various outcome measures of hematopoietic function), generally at ≥200 mg/kg-day following short-term (28-day), subchronic (90-day), or chronic (2-year) exposures.

[7] The "sufficient exposure conditions" are more fully evaluated and defined for the identified health effects through dose-response analysis in Section 5.

Table 3-25. Evidence profile table for hematopoietic effects

<table>
<tr><th colspan="5">Evidence stream summary and interpretation</th><th>Evidence integration summary judgment</th></tr>
<tr><td colspan="5">Evidence from studies of exposed humans</td><td rowspan="6">⊕⊕⊙
Evidence indicates (likely)

Primary basis:
Four high confidence studies in rats ranging from short term to chronic exposure durations, in both sexes, generally at ≥200 mg/kg-d

Human relevance:
Some species differences between human and rodent hematology exist, but these are not interpreted to impact judgments on the collection of rodent findings specific to PFHxA.

Cross-stream coherence:
N/A (human evidence indeterminate)

Susceptible populations and lifestages:
No evidence to inform</td></tr>
<tr><td>Studies and confidence</td><td>Factors that increase certainty</td><td>Factors that decrease certainty</td><td>Summary and key findings</td><td>Evidence stream judgment</td></tr>
<tr><td colspan="4">• There were no informative human studies available from the PFHxA evidence base.</td><td>⊙⊙⊙
Indeterminate</td></tr>
<tr><td colspan="5">Evidence from animal studies</td></tr>
<tr><td>Studies and confidence</td><td>Factors that increase certainty</td><td>Factors that decrease certainty</td><td>Summary and key findings</td><td>Evidence stream judgment</td></tr>
<tr><td>Hematology
4 high confidence studies in adult rats:
• 28-d
• 90-d (2 studies)
• 2-yr</td><td>• Consistent changes (decreases in hematocrit, hemoglobin, red blood cells, and MCHC and increases in reticulocytes, MCV, and MCH) across studies
• Coherence of red blood cells, HCT, and HGB and reticulocytes
• Large magnitude of effect as high as 356% for reticulocytes
• High confidence studies</td><td>• No factors noted</td><td>• Decreased red blood cells, hematocrit, and hemoglobin at ≥62.5 mg/kg-d; both sexes
• Increased MCH and MCV at ≥250; males more sensitive
• Increased reticulocytes at ≥200 mg/kg-d; both sexes, all studies
• Coherence of red blood cells and reticulocytes with splenic extramedullary hematopoiesis and bone marrow erythroid hyperplasia</td><td>⊕⊕⊕⊕
Robust

Findings considered adverse based on coherent evidence that was consistent across multiple laboratories and experimental designs. Consistent findings of decreased red blood cells, hematocrit, and hemoglobin at ≥200 mg/kg-d coherent with erythroid cell regeneration</td></tr>
</table>

Evidence stream summary and interpretation					Evidence integration summary judgment
Hemostasis 4 *high* confidence studies in adult rats: • 28-d • 90-d (2 studies) • 2-yr	• *Consistent* treatment related effect on platelet levels • *Consistency* across study designs • *High* confidence studies	• No factors noted	• Increased platelet levels ≥10 mg/kg-d; both sexes, 1 28-d, 2 90-d studies • Decreased activated partial thromboplastin (APTT) at ≥20 mg/kg-d; males only, 1 90-d study • Decreased prothrombin (PT) time at 500 mg/kg-d; males only, 1 90-d study		
Mechanistic evidence and supplemental information					
Biological events or pathways	**Species or model systems**	**Key findings, limitations, and interpretation**		**Evidence stream summary**	
• No informative studies identified.					

3.2.5. Endocrine Effects

Human

Thyroid Hormones

Two studies examined the association between PFHxA exposure and thyroid hormones in humans (see Figure 3-17). One was considered *uninformative* due to lack of consideration of confounding, including age, sex, and other factors which is expected to substantially impact the results (Seo et al., 2018). The other study was a cross-sectional study of the general population in China and was considered *low* confidence (Li et al., 2017) due to concerns around participant selection, outcome measures, consideration of confounding, and decreased sensitivity. Regarding the latter concern, the exposure levels were low and contrast narrow in Li et al. (2017) (median [range]: 0.01 [<LOD–1.1]) and 47% of samples were below the LOD. Among participants without thyroid disease, inverse associations with free T3 and thyroid stimulating hormone (TSH) were reported, with TSH being statistically significant (Pearson correlation coefficient = −0.27, $p < 0.01$). No association was observed with free T4. Total T4 and T3 were not measured in this study.

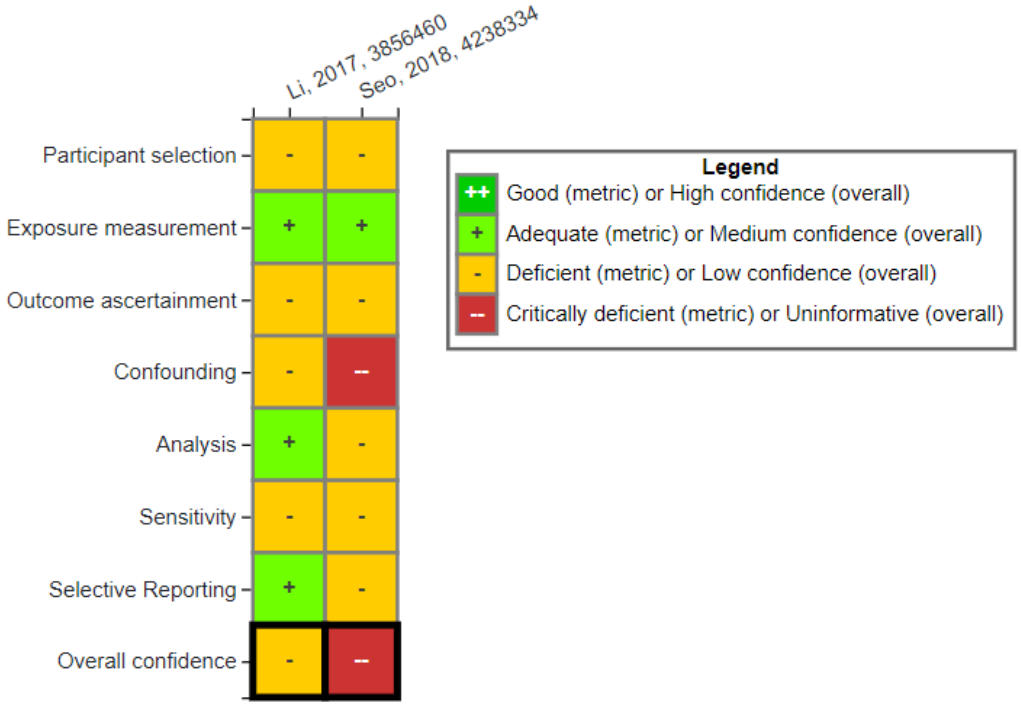


Figure 3-17. Study evaluation for human epidemiologic studies reporting toxicity findings from PFHxA exposures (HAWC: PFHxA – Human Toxicity Endocrine Effects link).

Animal

Four short-term (28-day), subchronic, and chronic animal studies evaluated potential endocrine effects of PFHxA or PFHxA sodium salt in rats. Most of the outcome-specific study ratings were rated *high* confidence. Histopathology for Chengelis et al. (2009b) was rated *low* confidence because of issues related to observational bias, concerns about endpoint sensitivity and specificity, and results presentation. A summary of the studies and the interpretations of confidence in the results for the different outcomes based on the individual study evaluations is presented in Table 3-26, and details are available by clicking the HAWC link.

Table 3-26. Endocrine endpoints for PFHxA and associated confidence scores from repeated-dose animal toxicity studies

Author (year)	Species, strain (sex)	Exposure design	Exposure route and dose range	Organ weight	Histopathology	Thyroid hormones
NTP (2018)	Rat, Harlan Sprague-Dawley (male and female)	Short term (28 d)	Gavage[a] Male and female: 0, 62.5, 125, 250, 500, 1,000 mg/kg-d	++	++	++
Chengelis et al. (2009b)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	Subchronic (90 d)	Gavage[a] Male and female: 0, 10, 50, 200 mg/kg-d	++	–	NM
Loveless et al. (2009)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	Subchronic (90 d)	Gavage[b] Male and female: 0, 20, 100, 500 mg/kg-d	++	++	NM
Klaunig et al. (2015)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	2-yr cancer bioassay	Gavage[a] Male: 0, 2.5, 15, 100 mg/kg-d Female: 0, 5, 30, 200 mg/kg-d	NM	++	NM

++ Outcome rating of *high* confidence; – outcome rating of *low* confidence; NM, outcome not measured.

[a,b]Study evaluation for animal toxicological endocrine endpoints reported from studies with male and female rats receiving PFHxA[a] or PFHxA sodium salt[b] by gavage. Study evaluation details for all outcomes are available by clicking the HAWC link.

++ Outcome rating of *high* confidence; – outcome rating of *low* confidence; NM, outcome not measured.

Thyroid Hormones

A single study evaluated potential PFHxA effects on endocrine function specific to thyroid hormones in rats exposed for 28 days (NTP, 2018). Specifically, males showed statistically significant, dose-dependent decreases in thyroid hormones. These outcomes showed a clear dose-dependent pattern of effect with treated animals showing reductions of 25%–73% or 20%–58% for free or total T4, respectively. Smaller decreases in T3 in males also were observed (18%–29%), although the dose-dependence of this effect was less clear. No treatment-related changes were

observed for T3 or T4 in females or for TSH in either sex (NTP, 2018). Results are summarized in Figure 3-18 and Table 3-27.

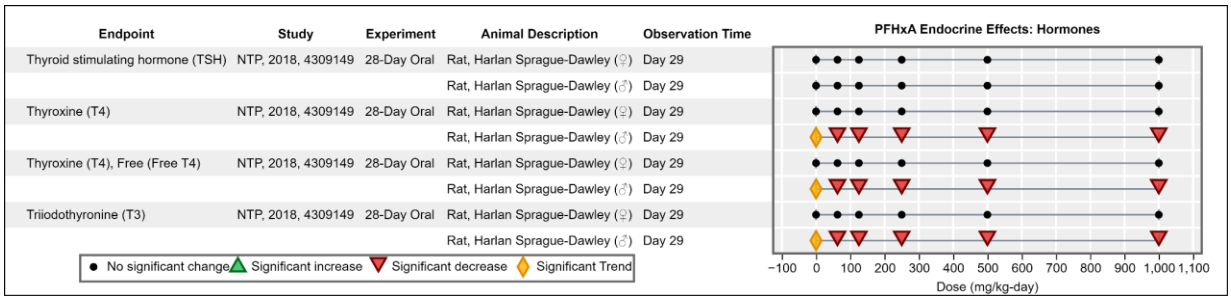


Figure 3-18. Thyroid hormone measures from the serum of rats exposed by gavage to PFHxA or PFHxA sodium salt (full details available by clicking the HAWC link).

Table 3-27. Percent change in thyroid hormone levels following PFHxA exposure in a 28-day oral toxicity study

Study design and reference	Hormone	Dose (mg/kg-d)				
		62.5	125	250	500	1,000
28-d, female rat (NTP, 2018)	Free T4	−1	−4	9	−4	−19
28-d, male rat (NTP, 2018)		−25*	−38*	−39*	−55*	−73*
28-d, female rat (NTP, 2018)	Total T4	−7	−11	−5	−9	−19
28-d, male rat (NTP, 2018)		−20*	−31*	−32*	−44*	−58*
28-d, female rat (NTP, 2018)	T3	−1	−6	3	14	−3
28-d, male rat (NTP, 2018)		−18*	−26*	−15*	−16*	−29*
28-d, female rat (NTP, 2018)	TSH	−15	−8	−9	40	−9
28-d, male rat (NTP, 2018)		9	5	6	9	−21

* Indicates instances where statistical significance ($p < 0.05$) compared to controls was reported by study authors.

Histopathology

Four studies evaluated histopathological changes in endocrine tissues, including the thyroid, pituitary, and pancreas, in rodents exposed to PFHxA (NTP, 2018; Klaunig et al., 2015; Chengelis et al., 2009b) or PFHxA sodium salt (Loveless et al., 2009). Of these, Loveless et al. (2009) reported thyroid follicular cell hypertrophy in both male and female rats exposed to PFHxA sodium salt for 90 days. The hypertrophy persisted after the exposure ceased, with females showing an increase at the 30-day (but not 90-day) recovery whereas, in males this effect was observed at the 90-day recovery time point. NTP (2018) reported this outcome was not affected by PFHxA following a 28-day exposure at doses as high as 1,000 mg/kg-day. The remaining two studies reported no treatment-related effects on thyroid histopathology at doses as high as 200 mg/kg-d

following subchronic (90-day) or chronic (2-year) exposure to PFHxA. Notably, Chengelis et al. (2009b) did not specify what outcomes were evaluated. Therefore, whether thyroid follicular cell hypertrophy was measured is unclear. No other treatment-related histopathological effects were noted in the PFHxA evidence base. Results are summarized in Table 3-28.

Table 3-28. Incidence of thyroid follicular epithelial cell hypertrophy following PFHxA or PFHxA ammonium salt exposures in rats

Sex (Timepoint)	0	2.5	5	10	15	20	30	50	62.5	100	125	200	250	500	1000
28-d exposure in rats (NTP, 2018)															
Female	0/10								0/10		0/10		0/10	0/10	0/10
Male	0/10								0/10		0/10		0/10	0/10	0/10
90-d exposure in rats (Loveless et al., 2009)															
Female (Exposure, Day 90)	0/10					0/10				0/11				4/10 *	
Male (Exposure, Day 90)	0/10					0/10				1/10 *				2/10 *	
Female (Recovery Day 30)	0/10													6/10 *	
Male (Recovery Day 30)	0/10													3/10 *	
Female (Recovery, Day 90)	0/10					0/10				0/9				0/10	
Male (Recovery, Day 90)	0/10					0/10				0/10				2/10 *	
2-yr exposure in rats (Klaunig et al., 2015)															
Female	0/21		0/25				0/20					0/14			
Male	0/18	0/24			0/23					0/24					

* Indicates instances where statistical significance ($p < 0.05$) compared to controls was reported by study authors; shaded cells represent doses not included in the individual studies.

Organ Weights

Three studies evaluated effects on thyroid and adrenal weights (NTP, 2018; Chengelis et al., 2009b; Loveless et al., 2009). Although no effects on relative thyroid weight were observed at the end of the 90-day exposure period in either subchronic study, Loveless et al. (2009) qualitatively reported a statistically significant increase in relative thyroid weight for female rats at the highest tested dose (500 mg/kg-day) of PFHxA sodium salt at the 30-day recovery. NTP (2018) observed a trend ($p < 0.05$) for decreased absolute adrenal gland weight in male rats exposed to 500 mg/kg-

day. No other treatment-related effects on endocrine organ weights were observed (NTP, 2018; Chengelis et al., 2009b; Loveless et al., 2009).

Mechanistic Evidence and Supplemental Information

Thyroid Hormone Levels in Zebrafish

(Zhang et al., 2022) evaluated the effects of a 96-hour exposure to PFHxA (0, 0.48, 2.4, and 12 mg/L) on whole body T3 and T4 levels in early-life stage zebrafish. Whole body total T3 showed a statistically significant increase in high dose animals. For T4 there was a significant dose-dependent reduction in at the low and mid dose, but a significant increase at the high dose.

Binding with transport proteins and nuclear receptor

Four studies were identified that investigated the ability of PFHxA to bind to two thyroid hormone serum transport proteins, transthyretin (Hamers et al., 2020; Ren et al., 2016; Weiss et al., 2009) and thyroid binding globulin (Ren et al., 2016), as well as the thyroid hormone receptor (Ren et al., 2015). All studies reported that PFHxA has a low binding affinity for transthyretin and the thyroid hormone receptors, and no detectable binding was observed for thyroid binding globulin.

Expression of HPT-related genes and proteins

Three studies evaluated the effects of PFHxA exposure on expression of mRNA and proteins related to thyroid function in early life-stage zebrafish (Zhang et al., 2022), embryonic avian neuronal cells (Vongphachan et al., 2011), and rat hepatoma cells (H4IIE). In zebrafish larvae exposed to up to 12 mg/L PFHxA for 96 h, whole body expression was increased for genes related to HPT regulation (*crh*, *trh*), thyroid hormone synthesis (*tg*, *nis*), transport (*ttr*), and nuclear receptors (*trα*, *trβ*) after PFHxA exposure. The same study also evaluated protein levels of thyroglobulin and transthyretin, which are important for thyroid hormone synthesis and transport, respectively, and reported treatment-related increases.

Two studies reported changes in mRNA expression of genes involved in metabolism of thyroid hormones (Zhang et al., 2022; Vongphachan et al., 2011). Deiodinases convert thyroid hormones to active and inactive forms through outer and/or inner ring deiodination. Differing patterns of expression were observed for deiodinases, with Vongphachan et al. (2011) finding an increase in deiodinase 2 and 3 in primary chicken neuronal cell cultures whereas Zhang et al. (2022) reported decreases in deiodinases 1 and 2 in early lifestage zebrafish. *Ugt1ab*, which mediates clearance of thyroid hormones from tissues, was decreased in whole body zebrafish larvae.

Naile et al. (2012) evaluated the effect of PFHxA on mRNA expression of genes related to thyroid development in rat hepatoma cells (H4IIE). Cells exposed to 0.1–100 μM for 72 hours showed increases (19.1–29.29-fold) in expression of *Hex*, which is important for cell differentiation during development of the thyroid. The effect on *Pax8*, a gene that regulates proliferation and

differentiation of cells that produce thyroxine, was less consistent. In general, there was an increase in expression (2.46-, 1.8-, and 3.73-fold for 0.1, 10, and 100 µM, respectively); however, there was a slight (0.92-fold) decrease in the 1 µm treated cells. The study did not report statistical significance of the results and the data are not sufficient for running statistical analyses as reported.

Evidence Integration

A single *low* confidence study provided some evidence of an association between PFHxA exposure and decreased T3 and TSH in humans, although methodological concerns reduce the reliability of these findings. Based on these results, there is *indeterminate* human evidence of endocrine effects.

Evidence supporting potential endocrine effects of PFHxA exposure is largely based on two *high* confidence rat studies showing decreases in serum thyroid hormone levels and increased thyroid epithelial cell hypertrophy in rats, but interpretation of these results is complex. The only available animal study that evaluated thyroid hormone levels showed a large magnitude of change in T4 (up to 73% decrease) and T3 (up to 20% decrease) with a clear dose-response for T4 (free and total) following exposure to PFHxA as low as 62.5 mg/kg-day, and these effects were observed only in males (NTP, 2018). A second study found increased incidence of thyroid epithelial cell hypertrophy following a 90-day exposure to PFHxA sodium salt (Loveless et al., 2009). Effects on thyroid hormone levels were also reported in larval zebrafish exposed to PFHxA during early development. Consistent with the available rodent data, whole body T4 levels were decreased, however T3 levels were increased in zebrafish which is the opposite direction of effects observed in the rat studies. The differences in effects on T3 may be explained by differences in the species (mammalian versus non-mammalian), methods (whole body levels versus serum), and lifestage (developing versus adult).

For the histopathological findings, treatment-related changes were reported for both males and females administered 500 mg/kg-day PFHxA sodium salt. The incidence of thyroid hypertrophy was higher in females than in males, although effects in males persisted longer after exposures had ceased. Also, no clear dose-response was found, with effects generally observed only at the highest dose tested. Three other studies evaluated thyroid histopathology but found no effects in either sex following a wide range of PFHxA exposure durations (28 days to 2 years) and doses (up to 1,000 mg/kg-day) (NTP, 2018; Klaunig et al., 2015; Chengelis et al., 2009b). No clear pattern of treatment-related effects was reported for endocrine organ weights.

Although the only available animal study examining thyroid hormones showed strong effects on T4 and T3 after short-term exposure, no effects were observed on TSH. The observed pattern of effects on the thyroid (i.e., decreased total and free T4 without a compensatory increase in TSH) after PFHxA exposure is consistent with thyroid perturbations following exposure to other PFAS, including PFBA (U.S. EPA, 2022b) and PFBS (U.S. EPA, 2021b). Rodents and humans share many similarities in the production, regulation, and functioning of thyroid hormones. Although differences exist, including the timing of in utero thyroid development and hormone turnover rates,

rodents are considered a good model for evaluating the potential for thyroid effects in humans (Zoeller et al., 2007). While there is uncertainty in the reliability of the TSH measurements and patterns of TH changes in animals may not translate perfectly to human clinical definitions, the observed decreases in total or free T4 in the absence of increases in TSH are considered biologically relevant to humans (Crofton, 2004; Lau et al., 2003). TSH is an indicator that the thyroid system has been perturbed, but it does not always change when serum T4 is decreased (Hood et al., 1999) and decreases in T4 alongside normal levels of TSH is consistent with the human clinical condition referred to as hypothyroxinemia [see additional discussion in (U.S. EPA, 2018b)]. Adverse neurological outcomes have been demonstrated following decreased T4 levels during the early neonatal period with no changes in T3 or TSH (Crofton, 2004). During pregnancy and early development, even transient perturbations in thyroid function can have permanent impacts on normal growth and neurodevelopment in the offspring. Although currently available evidence on thyroid hormones is limited to effects in males, there is evidence to support effects in females. Changes in thyroid histopathology were observed in both sexes, with a higher incidence in females (Loveless et al., 2009). Given the potential consistency of these findings with those observed for other PFAS, the availability of a single short-term study of thyroid hormones represents a significant data gap for PFHxA. The lack of consistency of the histopathology findings also reduces the strength of the available evidence; however, this variability across studies could be driven by differences in the dose levels examined (i.e., high dose ranged from 100 to 1,000 mg/kg-day) and exposure duration (i.e., short-term, subchronic, and chronic exposures).

While males exhibited a clear dose-dependent reduction in T4 and T3 at doses as low as 62.5 mg/kg-day in males but no effect in females at doses as high as 1000 mg/kg-day. These disparate results may be explained by sex-specific differences in the pharmacokinetics of PFHxA and its effect on induction of metabolizing enzymes. As discussed in Section 3.1.4, some evidence suggests sex-specific differences in the pharmacokinetics of PFHxA, with plasma concentrations measured 2–3 times higher in male rats when compared to females (Chang et al., 2008; Lau et al., 2006; Lau et al., 2004). Additionally, thyroid hormones may be reduced through PPARα activation and induction of microsomal enzyme inducers, such as CAR and PXR (Vansell, 2022). Males are more sensitive to PFHxA-induced PPARa activation (see Section 3.2.1). While these differences might explain why treatment-related effects on thyroid hormones were observed only in male rats, it is unclear why a similar sex-specific pattern was not observed for the reported thyroid histopathological changes observed at similar or higher doses.

Several mechanisms are implicated in the disruption of thyroid homeostasis. At present there is insufficient evidence identify the MOA(s) by which PFHxA induces the observed thyroid effects, but the mechanistic and supplemental information provide provides some support for PFHxA-related effects on thyroid function. PFHxA may affect thyroid hormone metabolism and bind to transport proteins and the nuclear receptor. Two mechanistic studies show effects on mRNA expression of deiodinases, which are important for conversion of thyroid hormones between active

and inactive forms (Zhang et al., 2022; Vongphachan et al., 2011) and glucuronidases which are important for excretion. Additionally, there is evidence that some PFAS may alter thyroid function via interaction with thyroid hormone receptors and transport proteins; however, the current data show only weak binding for PFHxA (Hamers et al., 2020; Ren et al., 2016; Ren et al., 2015; Weiss et al., 2009). Based on available data, there is *moderate* animal evidence of endocrine effects. In the absence of evidence to the contrary, effects in rats are considered relevant to humans.

Overall, the currently available ***evidence indicates*** that PFHxA likely causes endocrine effects in humans given sufficient exposure conditions (see Table 3-29).[8] This conclusion is based on four animal studies generally rated as *high* confidence that reported treatment-related decreases in thyroid hormone levels and more uncertain evidence of thyroid histopathology after exposure to PFHxA at ≥62.5 mg/kg-day.

[8] The "sufficient exposure conditions" are more fully evaluated and defined for the identified health effects through dose-response analysis in Section 5.

Table 3-29. Evidence profile table for endocrine effects

Evidence stream summary and interpretation					Evidence integration judgment
Evidence from studies of exposed humans					⊕⊕⊙ ***Evidence indicates (likely)*** *Primary basis:* Two animal studies generally rated as high confidence that reported treatment related changes in thyroid hormone levels, thyroid histopathology after exposure to PFHxA at ≥63.5 mg/kg-d. *Human relevance:* Without evidence to the contrary, effects in rats are considered relevant to humans.
Studies and confidence	**Factors that increase certainty**	**Factors that decrease certainty**	**Summary and key findings**	**Evidence stream judgment**	
Thyroid Hormones 1 *low* confidence study	• No factors noted	• *Lack of coherence* across related thyroid hormone measures • *Low* confidence study	• Inverse associations between free T3 and TSH and PFHxA in a single *low* confidence study	⊙⊙⊙ ***Indeterminate***	
Evidence from animal studies					
Studies and confidence	**Factors that increase certainty**	**Factors that decrease certainty**	**Summary and key findings**	**Evidence stream judgment**	
Thyroid Hormones 1 *high* confidence study in adult rats: • 28-d	• *High* confidence study. • *Dose response* gradient observed for free and total T4 • Large *effect magnitude*; up to 73%	• No factors noted	• Decreased T4 (free and total) and T3 observed in males only at ≥62.5 mg/kg-d	⊕⊕⊙ ***Moderate*** Some evidence of thyroid effects based on hormone and histopathological changes in two rat studies. Thyroid-related effects in	

Evidence stream summary and interpretation					Evidence integration judgment
Histopathology 3 *high* confidence studies in adult rats: • 28-d • 90-d • 2-yr 1 *low* confidence study in adult rats: • 90-d	• *High* confidence studies	• Unexplained inconsistency across studies	• Increased incidence of thyroid epithelial cell hypertrophy at ≥100 mg/kg-d for 90 d; persisted up to 90 d after exposure • No effects observed in 28 d study at up to 1,000 mg/kg-d	rodents are supported by mechanistic and supplemental information showing changes in thyroid hormone levels in zebrafish larvae, weak binding to thyroid transport proteins/receptor, and changes in thyroid related mRNA/protein expression.	*Cross-stream coherence:* N/A (human evidence *indeterminate*). *Susceptible populations and lifestages:* No evidence to inform *Other inferences:* Some mechanistic data and supplemental information were identified on this health effect were identified that provide additional support for potential PFHxA mediated endocrine effects but are insufficient to inform a potential MOA. Notably, the pattern of the
Organ Weight *High* confidence: 3 *high* confidence studies in adult rats: • 28-d • 90-d (2 studies)	• *High* confidence studies	• Unexplained inconsistency across studies	• Relative thyroid weights were increased only in females 30 d after exposure • Right adrenal weights decreased but no other adrenal effects were reported		

Evidence stream summary and interpretation			Evidence integration judgment
Mechanistic evidence and supplemental information			limited human and animal findings for PFHxA are consistent with hypothyroxinemia seen for other PFAS
Biological events or pathways	**Key findings, interpretation, and limitations**	**Evidence stream summary**	
Thyroid Hormones	*Key findings and interpretation:* • Altered whole body thyroid hormone levels (↑T3, ↓T4) in zebrafish larvae • Effects T3 shows opposite direction of effect of what was observed in both rodents and humans *Limitations:* Small evidence base with some inconsistencies in the pattern of effect observed in human and rodent data	Some support for potential effects of PFHxA on thyroid function.	
Receptor/ Transport Protein Binding	*Key findings and interpretation:* • Low binding potency for transthyretin and thyroid hormone receptor • No observed binding to thyroid binding globulin *Limitations:* Small evidence base showing weak effect.		
mRNA/ Protein Expression	*Key findings and interpretation*: • Altered expression of several thyroid-related mRNA and proteins observed in vivo (larval zebrafish) and in vitro (primary chicken neuronal cells; rat hepatoma, H4IIE) • Potential targets included pathways relevant to all aspects of thyroid regulation including, synthesis, release, transport, and metabolism of thyroid hormones as well as regulation of thyroid development, but in general each target evaluated by a single study • Lack of consistency of effect in two studies that evaluated mRNA expression of deiodinase 2 *Limitations*: Small evidence base lacking consistency across different studies (when available) for deiodinase 2		

3.2.6. Male Reproductive Effects

Human

Sperm Parameters

One *low* confidence study (Song et al., 2018) examined the association between PFHxA exposure and semen parameters and reported no decrease in concentration or motility with higher levels of PFHxA in serum (see Figure 3-19). A significant negative correlation between PFHxA levels in semen and sperm motility was found in this study (correlation coefficient = −0.35, $p < 0.01$), but analytical measurement of PFAS in semen is less established than in blood and the relevance to exposure is unclear. Still, exposure levels in the study based on serum measurements were high (median: 29 ng/mL, 5th–95th percentile: 11–70 ng/mL), so the study had reasonable sensitivity to detect an effect.

Reproductive Hormones

A single study rated *low* confidence due to low sensitivity and high potential for confounding (see Figure 3-19) found some support for associations between PFHxA and reproductive hormones in a population of adolescent (13–15 years old) males in Taiwan (Zhou et al., 2016). Overall, authors reported no significant associations between PFHxA and serum testosterone and estradiol; however, when the data were stratified by sex, a significant negative association between testosterone and PFHxA exposure level (β = −0.31, 95% CI: −0.59, −0.02) was found in boys. Based on serum measurements, the exposure levels in this study were low and the range narrow (median: 0.2 ng/mL, IQR 0.1–0.3 ng/mL), which might have reduced study sensitivity. The presence of an association despite reduced sensitivity could be due to either high potency of the exposure to cause these effects or potential confounding by other correlated PFAS, including PFOS, PFDA, and PFNA.

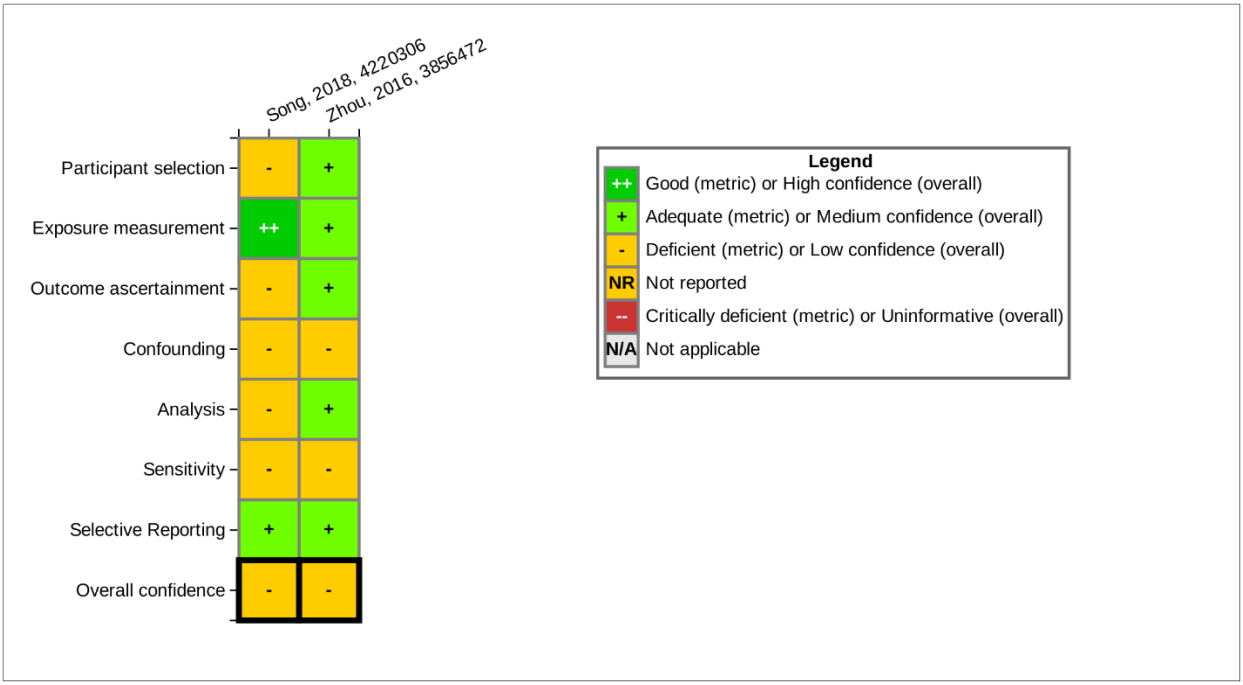


Figure 3-19. Study evaluation for human epidemiological studies reporting male reproductive findings from PFHxA exposures (HAWC: PFHxA – Human Toxicity Male Reproductive Effects link).

Animal

Several short-term (28-day), subchronic, and chronic animal studies evaluated sperm parameters, reproductive organ weights, and other reproductive male outcomes in rats receiving oral exposures of PFHxA and PFHxA sodium salt. Most outcome-specific study ratings were rated *high* confidence; however, some specific concerns were identified that resulted in *low* confidence ratings. Although generally a well-conducted study, NTP (2018) was rated *low* confidence for sperm parameters due to issues related to exposure duration and concerns for potential insensitivity. Histopathological results for Chengelis et al. (2009b) were rated *low* confidence because of issues related to observational bias, concerns about endpoint sensitivity and specificity, and results presentation. The results of the outcome-specific study evaluations are presented in Table 3-30, and details are available by clicking the HAWC link.

Table 3-30. Study design, exposure characteristics, and individual outcome ratings

Study	Species, strain (sex)	Exposure design	Exposure route and dose	Sperm parameters	Organ weight	Histopathology	Hormone levels	Reproductive system development
NTP (2018)	Rat, Harlan Sprague-Dawley (male and female)	Short term (28 d)	Gavage[a] Male and female: 0, 62.5, 125, 250, 500, 1,000 mg/kg-d	–	++	++	++	NM
Chengelis et al. (2009b)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	Subchronic (90 d)	Gavage[a] Male and female: 0, 10, 50, 200 mg/kg-d	NM	++	–	NM	NM
Loveless et al. (2009)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	Subchronic (90 d) One-generation reproductive: P_0 females dosed 70 d prior to cohabitation, through gestation and lactation (126 d); P_0 males dosed for 110 d Developmental: Gestation Days 6–20	Gavage[b] Male and female: 0, 20, 100, 500 mg/kg-d	++	++	++	NM	++
Klaunig et al. (2015)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	2-yr cancer bioassay	Gavage[a] Male: 0, 2.5, 15, 100 mg/kg-d Female: 0, 5, 30, 200 mg/kg-d	NM	NM	++	++	NM
Iwai and Hoberman (2014)[c]	Mouse, Crl:CD1(ICR); Charles River Laboratories, Inc.	Gestation Days 6–18	Gavage[d] Phase 1: 0, 100, 350, 500 mg/kg-d Phase 2: 0, 7, 35, 175 mg/kg-d	NM	NM	NM	NM	++

++ Outcome rating of *high* confidence; – outcome rating of *low* confidence; NM, outcome not measured.

[a,b,d]Study evaluation for animal toxicological endpoints reported from male reproductive studies with rats receiving PFHxA,[a] PFHxA sodium salt,[b] or PFHxA ammonium salt[d] by gavage. Study evaluation details for all outcomes are available by clicking the HAWC link.

[c]Phase 1 was a range-finding study used to determine the appropriate dose range for identification of a NOAEL in Phase 2.

Sperm Parameters

Evidence from a 28-day (NTP, 2018) and one-generation reproductive study (Loveless et al., 2009) included sperm parameters useful in evaluating potential male reproductive effects (see

Figure 3-20). In male rats receiving PFHxA daily by gavage for 28 days, a trend ($p < 0.05$) for decreased sperm count in the cauda epididymis was identified with a significant (25% change from control) decrease in the 1,000 mg/kg-day dose group. Animals in this dose group showed a significant decrease in body weight (13% change from control) at the end of the study but no other overt toxicity was indicated (e.g., mortalities or significant clinical observations) (NTP, 2018). Notably, these effects were observed despite concerns about sensitivity due to the short exposure duration of the study by NTP (2018) which does not encompass a full 6-week spermatogenic cycle in rats. In the one-generation reproductive study, Loveless et al. (2009) found no treatment-related effects for sperm parameters following a 10-week premating exposure in male rats to PFHxA sodium salt at doses up to 500 mg/kg-day. Results are summarized in Figure 3-20.

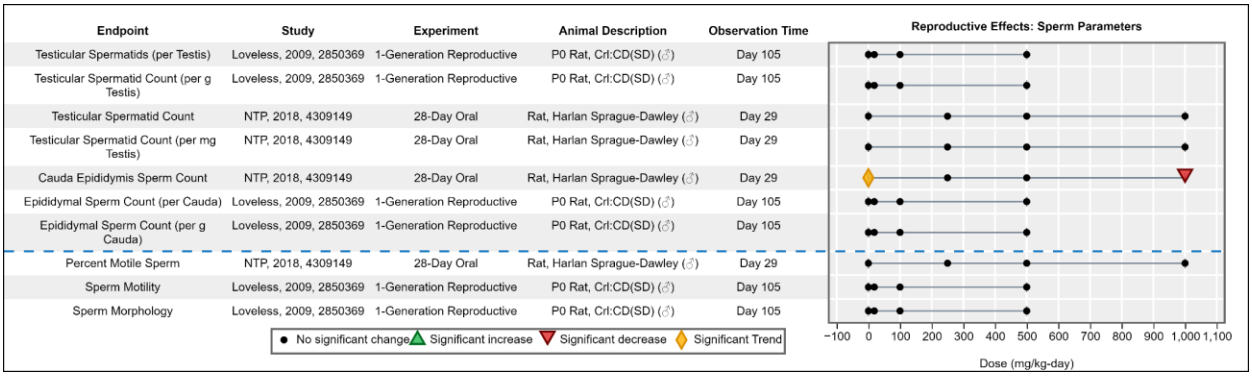


Figure 3-20. Male reproductive effects on sperm parameters in male rats exposed to PFHxA or sodium salt for 28 or 90 days (HAWC: PFHxA – Animal Toxicity Male Reproductive Effects link).

Reproductive Organ Weights

Reproductive studies commonly report both absolute and relative organ weights; however, for the testes, absolute weights are considered most informative for hazard evaluation (Bailey et al., 2004). Three studies (28- or 90-day exposure durations) reported data on the effects of PFHxA or PFHxA sodium salt exposure on male reproductive organ weights (i.e., testes, epididymis) in rats (see Figure 3-21) (NTP, 2018; Chengelis et al., 2009b; Loveless et al., 2009). Two studies reported a modest, but statistically significant ($p < 0.05$; 13%–16% change from control), increase in relative, but not absolute, testis weight in rats exposed to 1,000 mg/kg-day for 28 days (NTP, 2018) or 500 mg/kg-day for 90 days (Loveless et al., 2009). No treatment-related effects on male reproductive organ weights were reported by Chengelis et al. (2009b).

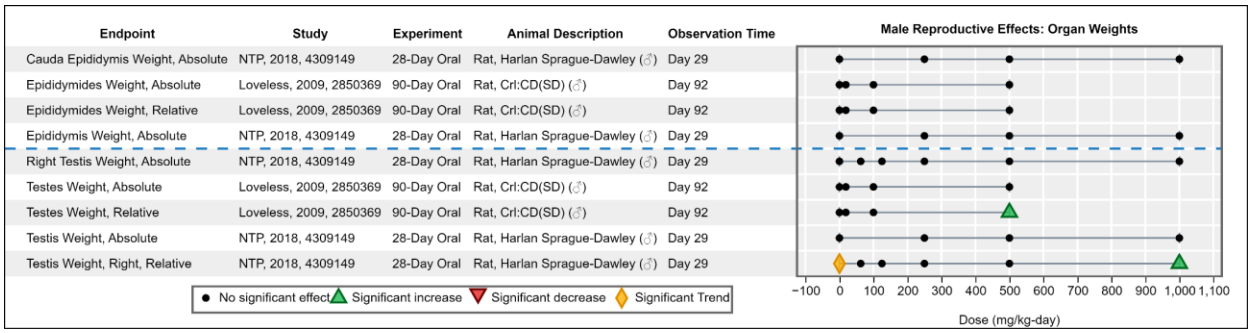


Figure 3-21. Male reproductive effects on epididymis and testis weight in rats exposed to PFHxA or PFHxA sodium salt (HAWC: PFHxA – Animal Toxicity Male Reproductive Effects link).

Reproductive Hormones

Two studies measured hormone levels (i.e., testosterone, estradiol, and luteinizing hormone) following exposure to PFHxA (NTP, 2018; Klaunig et al., 2015). Klaunig et al. (2015) reported a small, transient decrease in testosterone and luteinizing hormone in males at the 26-week time point. Effects were not dose dependent and were not significantly different from controls at doses up to 100 mg/kg-day PFHxA. This pattern was not observed at the 52-week time point. A short-term study found no effects on testosterone following exposure of up to 1,000 mg/kg-day for 28 days (NTP, 2018). Klaunig et al. (2015) also measured estradiol but found no treatment-related changes.

Histopathology

Four studies evaluated effects of PFHxA or PFHxA sodium salt on histopathology of the testes and epididymites and reported no treatment-related changes (NTP, 2018; Klaunig et al., 2015; Chengelis et al., 2009b; Loveless et al., 2009). One study was rated *low* confidence for this outcome (Chengelis et al., 2009b).

Male Reproductive System Development

Two studies examined outcomes related to male reproductive system development following developmental exposure to PFHxA ammonium or sodium salts (Iwai and Hoberman, 2014; Loveless et al., 2009). No treatment-related effects were reported on the age at preputial separation, a marker of puberty onset.

Evidence Integration

The available evidence informing the potential for an effect of PFHxA exposure on male reproduction in humans was limited to two *low* confidence studies that provided some indication of an association between PFHxA exposure and sperm motility (Song et al., 2018) and reproductive hormone levels (Zhou et al., 2016). These results are difficult to interpret, however, based on the

availability of a single study for each outcome and the high risk for bias in these evaluations. Based on these results, there is *indeterminate* human evidence of male reproductive effects.

In animals, the evidence supporting potential effects of PFHxA exposure on male reproduction was primarily limited to decreased sperm count (NTP, 2018) and increased relative testis weights (NTP, 2018; Loveless et al., 2009) at the highest tested doses in these studies (1,000 and 500 mg/kg-day, respectively). Decreased sperm count reported by NTP (2018) was considered *low* confidence due to the 28-day exposure duration and concerns that such short exposures would not capture the full spermatogenic cycle. Although finding effects in the presence of an insensitive exposure duration could indicate a sensitive window for chemical-specific perturbations, similar results were not observed in a *high* confidence subchronic study performed in the same rat strain (Loveless et al., 2009), albeit the highest tested dose was 500 as compared to 1,000 mg/kg-day in the short-term study. In addition, evidence of overt toxicity (i.e., 13% reduction in terminal body weight relative to controls) was found in the male rats dosed 1,000 mg/kg-day in the NTP (2018) study.

Two studies reported increased relative testis weight; however, the preferred metric of absolute testis weight did not change in either study and no changes in organ weight were observed in a second subchronic study (Chengelis et al., 2009b). Reproductive hormone (i.e., testosterone and luteinizing hormone) levels were reduced in the only chronic study; however, the effect was small in magnitude, was not dose-dependent, and was observed only at the 26-week time point (Klaunig et al., 2015). Similar results on testosterone were not reported in the short-term *high* confidence study (NTP, 2018). No other coherent findings (i.e., reproductive histopathology and male reproductive system development) supporting reproductive toxicity were identified in the animal evidence base. Based on these results there is *indeterminate* animal evidence of male reproductive effects.

Overall, the currently available ***evidence is inadequate*** to assess whether PFHxA might cause male reproductive effects in humans (see Table 3-31).

Table 3-31. Evidence profile table for male reproductive effects

Evidence stream summary and interpretation					Evidence integration summary judgment
Evidence from studies of exposed humans					⊙⊙⊙ ***Evidence inadequate*** *Primary Basis:* Evidence is *low* confidence or largely null *Human relevance:* N/A (*indeterminate* animal evidence) *Cross stream coherence:* N/A (human evidence *indeterminate*) *Susceptible population and lifestages:* No evidence to inform
Studies and confidence	**Factors that increase certainty**	**Factors that decrease certainty**	**Summary and key findings**	**Evidence stream judgment**	
Sperm Parameters 1 *low* confidence study	• No factors noted	• Low confidence study.	• Association between PFHxA levels in semen and decreased sperm motility	⊙⊙⊙ ***Indeterminate***	
Reproductive Hormones 1 *low* confidence study	• No factors noted	• Low confidence study	• Significant inverse association between PFHxA exposure and testosterone despite poor sensitivity		
Evidence from animal studies					
Studies and confidence	**Factors that increase certainty**	**Factors that decrease certainty**	**Summary and key findings**	⊙⊙⊙ ***Indeterminate***	
Sperm Parameters 1 *high* confidence study in adult rats: • 90-d 1 *low* confidence in adult rats • 28-d	• No factors noted	• Unexplained inconsistency across studies	• Decreased sperm count in the cauda epididymis at 1,000 mg/kg-d	The data are largely null. Some evidence of reproductive effects but interpretation limited by unexplained inconsistency at effects observed only at the high dose that elicited high overt toxicity (i.e., 13% decrease in body weight).	
Organ Weights 3 *high* confidence studies in adult rats:	• *High* confidence studies	• No factors noted	• Increased relative testis weight at ≥500 mg/kg-d; no change in absolute		

Evidence stream summary and interpretation					Evidence integration summary judgment
• 28-d • 90-d (2 studies)	• *Dose response* with longer exposure duration		testis weights (preferred metric)		
Reproductive Hormones 2 *high* confidence studies in adult rats: • 28-d • 2-yr	• *High* confidence studies	• No factors noted	• Transient decrease of small magnitude in luteinizing hormone and testosterone		
Histopathology and Male Reproductive System Development 4 *high* confidence studies in rats and mice: • 28-d (rat) • 90-d (rat) • GD 6–18 (mouse) • 2-yr (rat) 1 *low* confidence study in adult rats: • 90-d	• *High* confidence studies	• No factors noted	• No treatment-related effects reported at ≤1,000 mg/kg-d		
Mechanistic evidence and supplemental information					
Biological events of pathways	**Biological events of pathways**	**Biological events of pathways**		**Biological events of pathways**	
• No studies identified					

3.2.7. Female Reproductive Effects

Human

Reproductive Hormones

A single *low* confidence study (see Figure 3-22) evaluated associations between PFHxA and reproductive hormones in a population of Taiwanese adolescents (13–15 years old) (Zhou et al., 2016). Overall, the authors reported nonsignificant inverse associations between PFHxA and serum testosterone and estradiol in females when the data were stratified by sex. Exposure levels to PFHxA were low, which might have reduced study sensitivity, as described above in Section 3.2.6. Male Reproductive Effects.

Figure 3-22. Study evaluation for human epidemiological studies reporting female reproductive findings from PFHxA exposures (HAWC: PFHxA – Human Toxicity Female Reproductive link).

Animal

Five animal studies evaluated outcomes related to female reproduction in rats and mice receiving PFHxA via gavage, PFHxA sodium salt, or PFHxA ammonium salt. Study designs included short-term (28-day), subchronic (90-day), and chronic (2-year) one-generation reproductive and developmental exposures. In general, the outcome-specific study ratings were *high* confidence. One study was rated *low* confidence for histopathology due to concerns about observational bias, endpoint sensitivity and specificity, and results presentation (Chengelis et al., 2009b). The results of

study evaluation for female reproductive outcomes are presented in Table 3-32 and details are available by clicking the HAWC link.

Table 3-32. Study design characteristics

Study	Species, strain (sex)	Exposure design	Exposure route and dose	Fertility and pregnancy	Organ weight	Histopathology	Reproductive hormones	Reproductive system development
NTP (2018)	Rat, Harlan Sprague-Dawley (male and female)	Short term (28 d)	Gavage[a] Male and female: 0, 62.5, 125, 250, 500, 1,000 mg/kg-d	++	++	++	++	NM
Chengelis et al. (2009b)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	Subchronic (90 d)	Gavage[a] Male and female: 0, 10, 50, 200 mg/kg-d	NM	++	–	NM	NM
Loveless et al. (2009)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	Subchronic (90 d) One-generation reproductive: P_0 females dosed 70 d prior to cohabitation, through gestation and lactation (126 d); P_0 males dosed for 110 d Developmental: Gestation Days 6–20	Gavage[b] Male and female: 0, 20, 100, 500 mg/kg-d	++	++	++	NM	++
Klaunig et al. (2015)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	2-yr cancer bioassay	Gavage[a] Male: 0, 2.5, 15, 100 mg/kg-d Female: 0, 5, 30, 200 mg/kg-d	NM	NM	++	++	NM
Iwai and Hoberman (2014)[c]	Mouse, Crl: CD1(ICR) () (male and female)	Developmental: Gestation Days 6–18	Gavage[d] Phase 1: 0, 100, 350, 500 mg/kg-d Phase 2: 0, 7, 35, 175 mg/kg-d	++	NM	++	NM	++

++ Outcome rating of *high* confidence; – outcome rating of *low* confidence; NM, outcome not measured.

[a,b]Study evaluation for animal toxicological endpoints reported from female reproductive studies with rats receiving PFHxA,[a] PFHxA sodium salt,[b] or PFHxA ammonium salt[d] by gavage. Study evaluation details for all outcomes are available by clicking the HAWC link.

[c]Phase 1 was a range-finding study used to determine the appropriate dose range for identification of a NOAEL in Phase 2.

++ Outcome rating of *high* confidence; – outcome rating of *low* confidence; NM, outcome not measured.

Fertility and Pregnancy Outcomes

Three studies published in two reports evaluated outcomes related to fertility and pregnancy following exposure by gavage with PFHxA or PFHxA salts in rats or mice (Iwai and

Hoberman, 2014; Loveless et al., 2009). Some effects on maternal body weight change (i.e., gain or loss) were noted. In both the developmental and one-generation reproductive rat studies (Loveless et al., 2009), statistically significant reductions in maternal body weight change were observed during gestation in the high dose group (500 mg/kg-day). In the developmental study (Loveless et al., 2009), there was a statistically significant decrease in total maternal body weight gain (19% relative to control) and when correcting for gravid uterine weight (26% relative to control) from GD 6–21 in the 500 mg/kg-day dose group. In the one-generation reproductive study, similar effects were observed but were limited to early gestation (Loveless et al., 2009). From GD 0–7, body weight gain in dams exposed to 500 mg/kg-day was reduced by 31% relative to controls. There was no treatment-related effect on maternal weight gain over the entire gestational period (GD 0–21) and the high dose (500 mg/kg-day) showed a statistically significant increase in body weight change relative to controls during lactation (PND 0–21) (Loveless et al., 2009). No changes in maternal body weight gain were identified in mice (Iwai and Hoberman, 2014).

Only one of the three available studies reported effects on absolute maternal body weight. In the developmental rat study, dams exposed to 500 mg/kg-day (GD 6–20) showed a statistically significant decrease in terminal body weight (7% relative to control) (Loveless et al., 2009). Deficits remained when correcting for gravid uterine weight (5% relative to control), indicating the effects on body weight were driven by maternal body weight rather than reductions in fetal body weight or number of fetuses. However, this level of change may not be biologically significant (U.S. EPA, 1991). There was no effect on absolute maternal body weight in the one-generation reproductive rat or mouse study (Iwai and Hoberman, 2014; Loveless et al., 2009). These results are presented in Figure 3-23.

No treatment-related effects on mating, pregnancy incidence, gestation length, number of implantations, or litter size were reported in either study that evaluated these outcomes (Iwai and Hoberman, 2014; Loveless et al., 2009). Estrous cyclicity in rats exposed as adults or during gestation was also unaffected in two studies (NTP, 2018; Loveless et al., 2009).

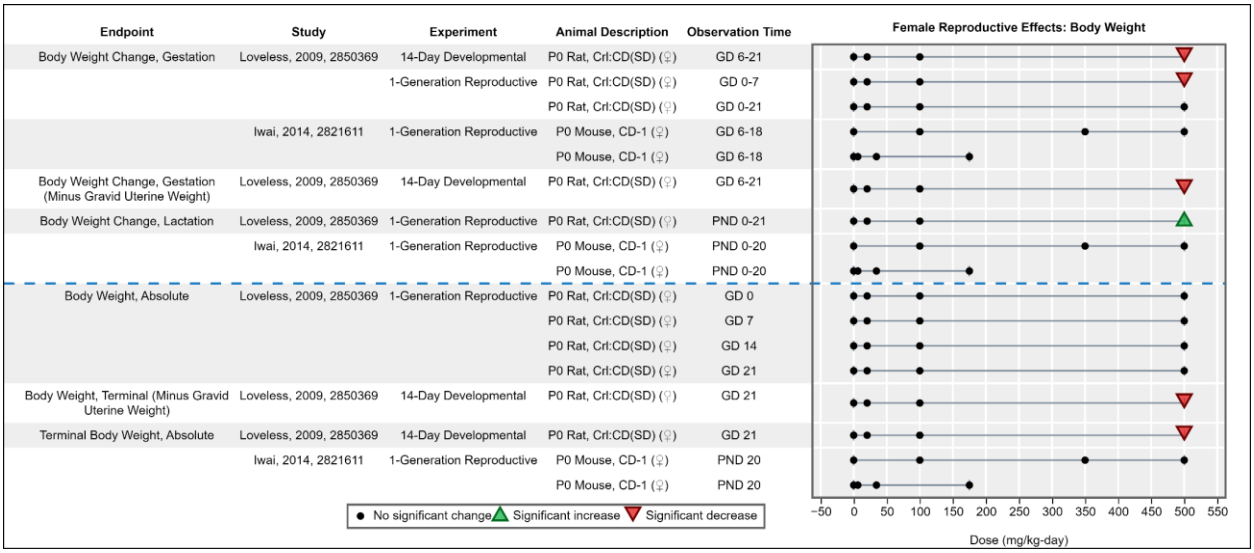


Figure 3-23. Effects on body weight in female rats and mice exposed to PFHxA or PFHxA ammonium salt in reproductive studies (HAWC: PFHxA – Animal Toxicity Female Reproductive Supporting Table).

Histopathology

Four studies evaluated effects on histopathology of reproductive organs (i.e., uterus and ovaries) in rodents following exposure to PFHxA (NTP, 2018; Klaunig et al., 2015; Chengelis et al., 2009b) or PFHxA sodium salt (Loveless et al., 2009). Only NTP (2018) reported an effect of exposure, with females showing a statistically significant increase in the incidence of bilateral uterine horn dilation in all but the vehicle controls and highest dose group (see Figure 3-24). Whereas the control and high-dose group had 10 animals per group, however, groups showing a statistically significant increase had only 1–3 animals per group, complicating interpretation of these findings. The total incidence ranges from 1 to 3 animals/treatment group, regardless of sample size or PFHxA dose (see Figure 3-24). The biological significance of these results is unclear. Uterine horn dilation can indicate an estrogenic effect, but no coherent changes in serum estradiol or estrous cyclicity were observed in this study. Similarly, no other treatment-related effects on female reproductive histopathology were reported (NTP, 2018; Klaunig et al., 2015; Chengelis et al., 2009b; Loveless et al., 2009).

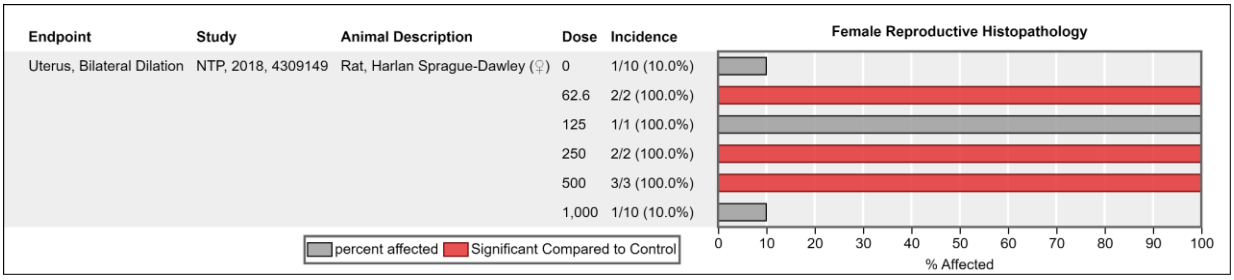


Figure 3-24. Female reproductive effects on uterine horn dilation in rats exposed to PFHxA for 28 days (HAWC: PFHxA – Animal Toxicity Female Reproductive link).

Organ Weights

Three studies evaluated effects of PFHxA exposure on uterine and ovarian weights (NTP, 2018; Chengelis et al., 2009b; Loveless et al., 2009). Authors reported no treatment-related effects for these outcomes.

Reproductive Hormones

Two studies measured effects of PFHxA or PFHxA ammonium salt on testosterone (NTP, 2018; Klaunig et al., 2015), estradiol, and luteinizing hormone (Klaunig et al., 2015). No treatment-related effects were reported in either study.

Female Reproductive System Development

Two studies evaluated the potential for reproductive development effects following developmental exposure to PFHxA ammonium or sodium salts. Iwai and Hoberman (2014) and Loveless et al. (2009) found no effects on age at vaginal opening, a measure of puberty onset.

Evidence Integration

A single *low* confidence human study reported a weak inverse association between PFHxA exposure measures and serum levels of reproductive hormone levels in adolescents (Zhou et al., 2016). Based on these results, there is *indeterminate* human evidence of female reproductive effects.

In animals, evidence supporting effects of PFHxA exposure female reproduction was largely limited to effects on maternal weight gain during gestation in rats exposed to 500 mg/kg-day (Loveless et al., 2009). These effects corresponded with a small but statistically significant absolute body weight in the *high* confidence developmental rat study only (Loveless et al., 2009), however the level of the decrease (5%–7%) may not be biologically significant. There were no effects on maternal weight or weight gain in the in the mouse study. The reported effects on uterine horn dilation appears to be influenced by differences in sample sizes, as the total incidence of the finding is similar across controls and all dosing groups. Furthermore, this finding is generally associated with estrogenic effects, but no coherent changes were observed that would be indicative of

estrogenic changes in females. No treatment-related changes were reported for other female reproductive outcomes (NTP, 2018; Klaunig et al., 2015; Iwai and Hoberman, 2014; Chengelis et al., 2009b; Loveless et al., 2009). Based on these results, there is *indeterminate* animal evidence of female reproductive effects.

Overall, the currently available ***evidence is inadequate*** to assess whether PFHxA might cause female reproductive effects in humans (see Table 3-33).

Table 3-33. Evidence profile table for female reproductive effects

Evidence stream summary and interpretation					Evidence integration summary judgment
Evidence from studies of exposed humans					⊙⊙⊙ ***Evidence inadequate***
Studies and confidence	**Factors that increase strength**	**Factors that decrease certainty**	**Summary and key findings**	**Evidence stream judgment**	
Reproductive Hormones 1 *low* confidence study	• No factors noted	• *Low* confidence study	• Nonsignificant inverse association between PFHxA exposure and testosterone and estradiol	⊙⊙⊙ ***Indeterminate***	
Evidence from animal studies					
Studies and confidence	**Factors that increase strength**	**Factors that decrease certainty**	**Summary and key findings**	**Evidence stream judgment**	*Primary Basis:* Evidence is *low* confidence or largely null. *Human relevance:* • N/A (human and animal evidence both *indeterminate*) *Cross stream coherence:* • N/A (human and animal evidence both *indeterminate*)
Fertility and Pregnancy Outcomes 3 *high* confidence studies in rats and mice: • 28-d (rat) • 90-d (rat) • GD 6–18 (mouse)	• *High* confidence studies	• Unexplained inconsistency across studies	• Decreases in maternal weight gain during gestation in rats exposed to 500 mg/kg-d	⊙⊙⊙ ***Indeterminate*** The animal evidence is largely null. Some evidence of female reproductive effects but body weight effects lacked consistency across studies. Histopathology effects were not dose-dependent and lacked coherent evidence to support the biological significance of the findings	
Histopathology 4 *high* confidence studies in rats and mice: • 28-d (rat)	• *High* confidence studies	• Unexplained inconsistency across studies • *Lack of expected coherence* with	• Increase in bilateral uterus dilation reported for all groups except the highest dose		

Evidence stream summary and interpretation					Evidence integration summary judgment
• 90-d (rat) • 2-yr (rat) • GD 6–18 (mouse) 1 *low* confidence study in adult rats: • 90-d		other estrogen related outcomes			*Susceptible populations:* • None identified
Organ Weights, Reproductive Hormones, Reproductive System Development 6 *high* confidence studies in rats and mice: • 28-d (rat) • 90-d (rat, 2 studies) • 2-yr (rat) • GD 6–18 (mouse) • GD 6–20 (rat)	• *High* confidence studies	• No factors noted	• No treatment-related effects were reported at ≤1,000 mg/kg-d		
Mechanistic evidence and supplemental information					
Biological events of pathways	**Biological events of pathways**	**Biological events of pathways**		**Biological events of pathways**	
• No studies Identified					

3.2.8. Immune Effects

Human

Asthma, Immune Markers, and Potentially Related Respiratory Outcomes

One *medium* confidence case-control study in Taiwan of asthma was reported in three publications (Qin et al., 2017; Zhou et al., 2017; Dong et al., 2013). Dong et al. (2013) includes results from all three studies that examined the potential association between PFHxA exposure and asthma incidence or severity, control of asthma symptoms and related immune markers (see Figure 3-25). The study also measured pulmonary function. The only finding of note was a nonmonotonic positive association between incident asthma (i.e., diagnosis in the previous year) and PFHxA exposure (odds ratio [95% CI] for Q2: 1.2 [0.7, 2.1], Q3: 0.9 [0.5, 1.6], Q4: 1.6 [0.9, 2.9]) that was not statistically significant. No clear association was found with asthma severity or control of asthma symptoms (Dong et al., 2013), pulmonary function measured with spirometry (Qin et al., 2017), or immune markers (Dong et al., 2013) among children with asthma. The exposure levels in this study were low and contrast narrow (median [IQR]: 0.2 ng/mL [0.1–0.3 ng/mL]), which may have reduced study sensitivity.

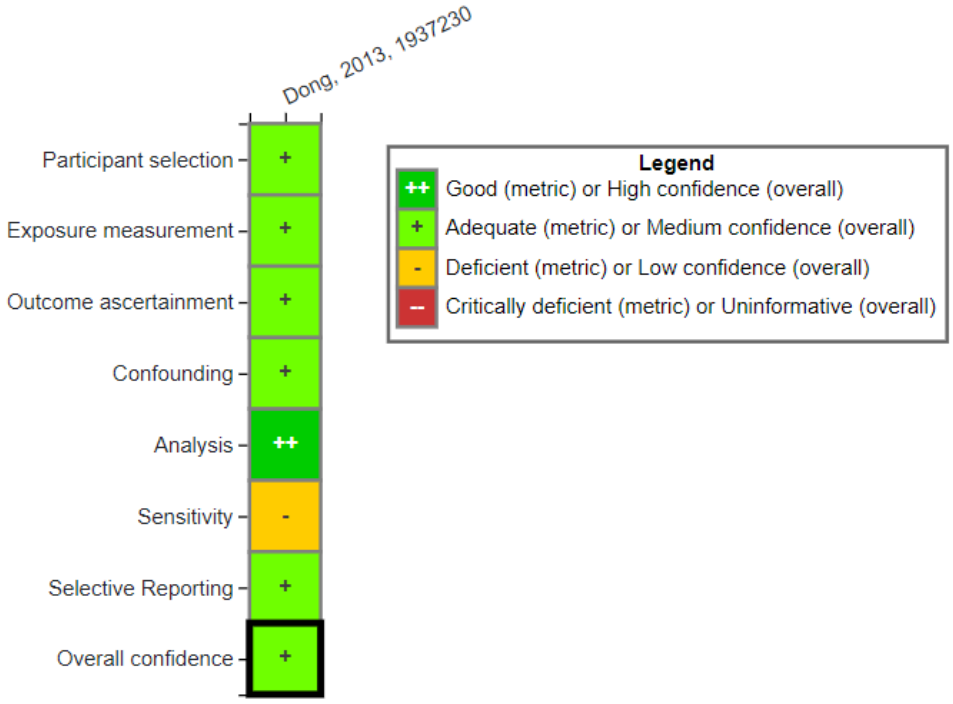


Figure 3-25. Study evaluation for human epidemiological studies reporting findings from PFHxA exposures (HAWC: PFHxA – Human Toxicity Immune Effects link).

The evaluation of Dong et al. (2013) encompasses all publications related to this study.

Animal

Several animal studies, including short-term (28-day), subchronic, reproductive, and designs, evaluated toxicological findings of immune effects in rats receiving oral exposures of PFHxA and PFHxA sodium salt. Most of the outcome-specific study ratings were considered *high* or medium confidence; however, some specific concerns were identified that resulted in a *low* confidence rating. Histopathology for Chengelis et al. (2009b) was rated *low* confidence because of issues related to observational bias, concerns about endpoint sensitivity and specificity, and results presentation. The results of the outcome-specific study evaluations are presented in Table 3-34 and details are available by clicking the HAWC link.

Table 3-34. Study design characteristics and individual outcome ratings for immune endpoints

Study	Species, strain (sex)	Exposure design	Exposure route and dose	Organ weight	Histopathology	Immune cell counts
NTP (2018)	Rat, Harlan Sprague-Dawley (male and female)	Short term (28 d)	Gavage[a] Male and female: 0, 62.5, 125, 250, 500, 1,000 mg/kg-d	++	++	++
Kirkpatrick (2005a)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	Reproductive: P_0 animals dosed 14 d prior to mating through end of mating period (male) or PND 4 (female)	Gavage[a] Male and female: 50, 150 and 300/450[c] mg/kg/d	+	+	+
Chengelis et al. (2009b)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	Subchronic (90 d)	Gavage[a] Male and female: 0, 10, 50, 200 mg/kg-d	++	–	++
Loveless et al. (2009)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	Subchronic (90 d)	Gavage[b] Male and female: 0, 20, 100, 500 mg/kg-d	++	++	++
Klaunig et al. (2015)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	2-yr cancer bioassay	Gavage[a] Male: 0, 2.5, 15, 100 mg/kg-d Female: 0, 5, 30, 200 mg/kg-d	NM	++	++

++ Outcome rating of *high* confidence; + outcome rating of *medium* confidence; – outcome rating of *low* confidence; NM, outcome not measured.

[a,b]Study evaluation for animal toxicological immune endpoints reported from studies with male and female rats receiving PFHxA[a] or PFHxA sodium salt[b] by gavage. Study evaluation details for all outcomes are available by clicking the HAWC link.

[c]Due to high toxicity at the 450 mg/kg-d exposed animals, dosage was reduced to 300 mg/kg-d on exposure day 4.

Organ Weights

Four studies evaluated effects on spleen and thymus weights in response to PFHxA (NTP, 2018; Chengelis et al., 2009b; Kirkpatrick, 2005a) or PFHxA sodium salt (Loveless et al., 2009) exposure.

The available evidence identified, in general, decreased absolute or relative thymus weights. Two studies reported statistically significant decreases in absolute thymus weights in males exposed to 500 mg/kg-day PFHxA sodium salt for 90 days (Loveless et al., 2009) or 450/300 mg/kg-day in reproductive study (Kirkpatrick, 2005a). Similarly, downward trends in both relative and absolute thymus weights thymus were reported in males and females receiving PFHxA in the short term (NTP, 2018). A single study qualitatively reported no treatment-related effects on thymus weights (Chengelis et al., 2009b).

Spleen weights did not show a clear pattern of effect across studies. In the short-term study, a trend of increased weights in males and females receiving PFHxA (NTP, 2018) was observed, whereas spleen weights were decreased in males receiving PFHxA sodium salt in the 90-day study by Loveless et al. (2009). Both Chengelis et al. (2009b) and Kirkpatrick (2005a) reported no treatment-related effects on spleen after exposure to ≤450 mg/kg-day PFHxA for up to 90 days. Results are summarized in Figure 3-26.

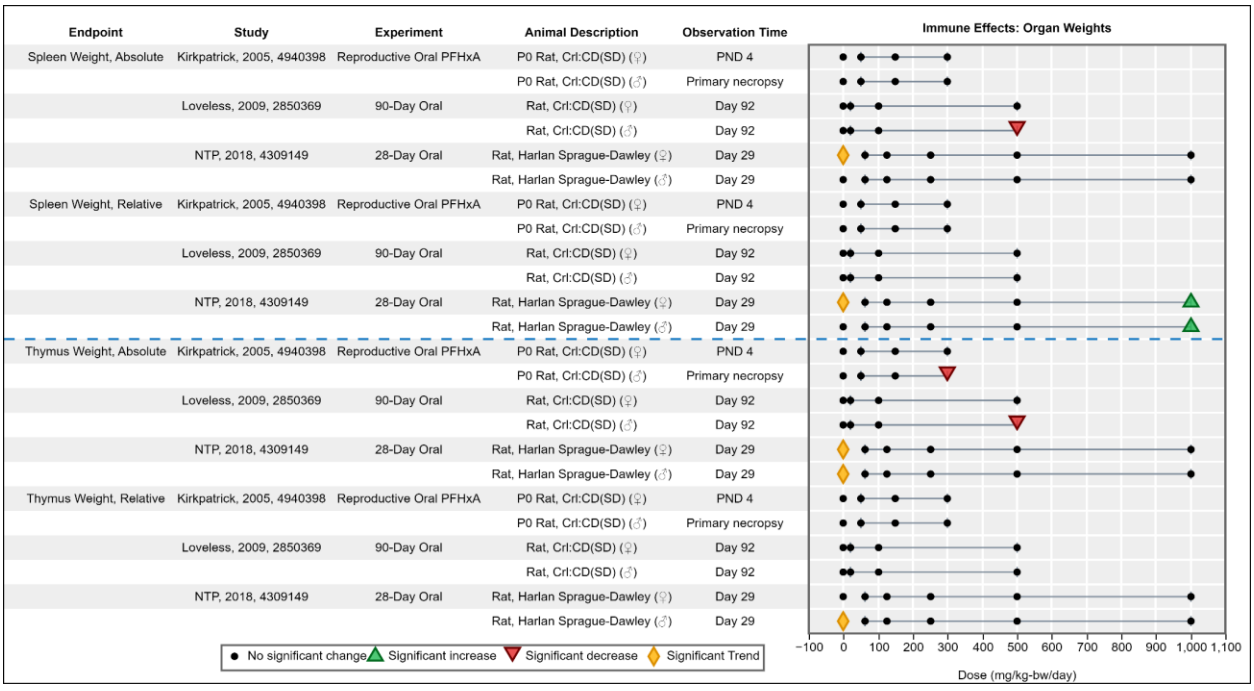


Figure 3-26. Immune organ weights in rats exposed by gavage to PFHxA or PFHxA sodium salt (HAWC: PFHxA – Animal Toxicity Immune Effects link).

Histopathology

Five studies examined spleen, thymus, lymph nodes, or bone marrow for histopathological changes (NTP, 2018; Klaunig et al., 2015; Chengelis et al., 2009b; Loveless et al., 2009; Kirkpatrick, 2005a). Some evidence of effects on immune related histopathology was reported by NTP (2018) reported an increased incidence of extramedullary hematopoiesis in the spleens of males and females at 1,000 mg/kg-day after a 28-day exposure. Minimal to mild extramedullary hematopoiesis also was found in the spleens of male rats receiving 500 mg/kg-day PFHxA sodium salt (Loveless et al., 2009). This effect was coincident with erythroid hyperplasia of the bone marrow of males and females and might be related to the effects on red blood cells (discussed in "Hemostasis" of Section 3.2.4) rather than an immune-specific effect. These changes did not persist after the 30-day recovery and specific incidence data were not reported (Loveless et al., 2009). A reproductive study in rats reported increased incidence of histopathologic changes in the spleen, thymus, and lymph nodes in the high dose group (450/300 mg/kg-day). Notably, these findings were limited to unscheduled deaths (i.e., found dead or euthanized in extremis) in animals showing signs of overt toxicity, therefore, these results may not reflect an immune-specific effect. No other effects were reported for immune-related tissues (NTP, 2018; Klaunig et al., 2015; Chengelis et al., 2009b; Loveless et al., 2009; Kirkpatrick, 2005a).

Immune Cell Counts

Five animal studies evaluated hematological indicators of immunotoxicity (NTP, 2018; Klaunig et al., 2015; Chengelis et al., 2009b; Loveless et al., 2009; Kirkpatrick, 2005a). Of these studies, NTP (2018) and Loveless et al. (2009) reported increased neutrophils at doses as low as 20 mg/kg-day and decreased basophils in males receiving ≥250 and 500 mg/kg-day PFHxA or PFHxA sodium salt, respectively. No effects were observed on basophils or neutrophils in the other three rat studies (reproductive, -d90-, and -y2-) at exposures to PFHxA as high as 450 mg/kg-day (Klaunig et al., 2015; Chengelis et al., 2009b; Kirkpatrick, 2005a). Eosinophils were decreased only in males exposed to PFHxA sodium salt for 90 days (Loveless et al., 2009). No other treatment-related effects were reported for specific white blood cell populations or total white blood cell counts following PFHxA or PFHxA sodium salt exposures in rats (NTP, 2018; Klaunig et al., 2015; Chengelis et al., 2009b; Loveless et al., 2009; Kirkpatrick, 2005a). Results are summarized in Figure 3-27.

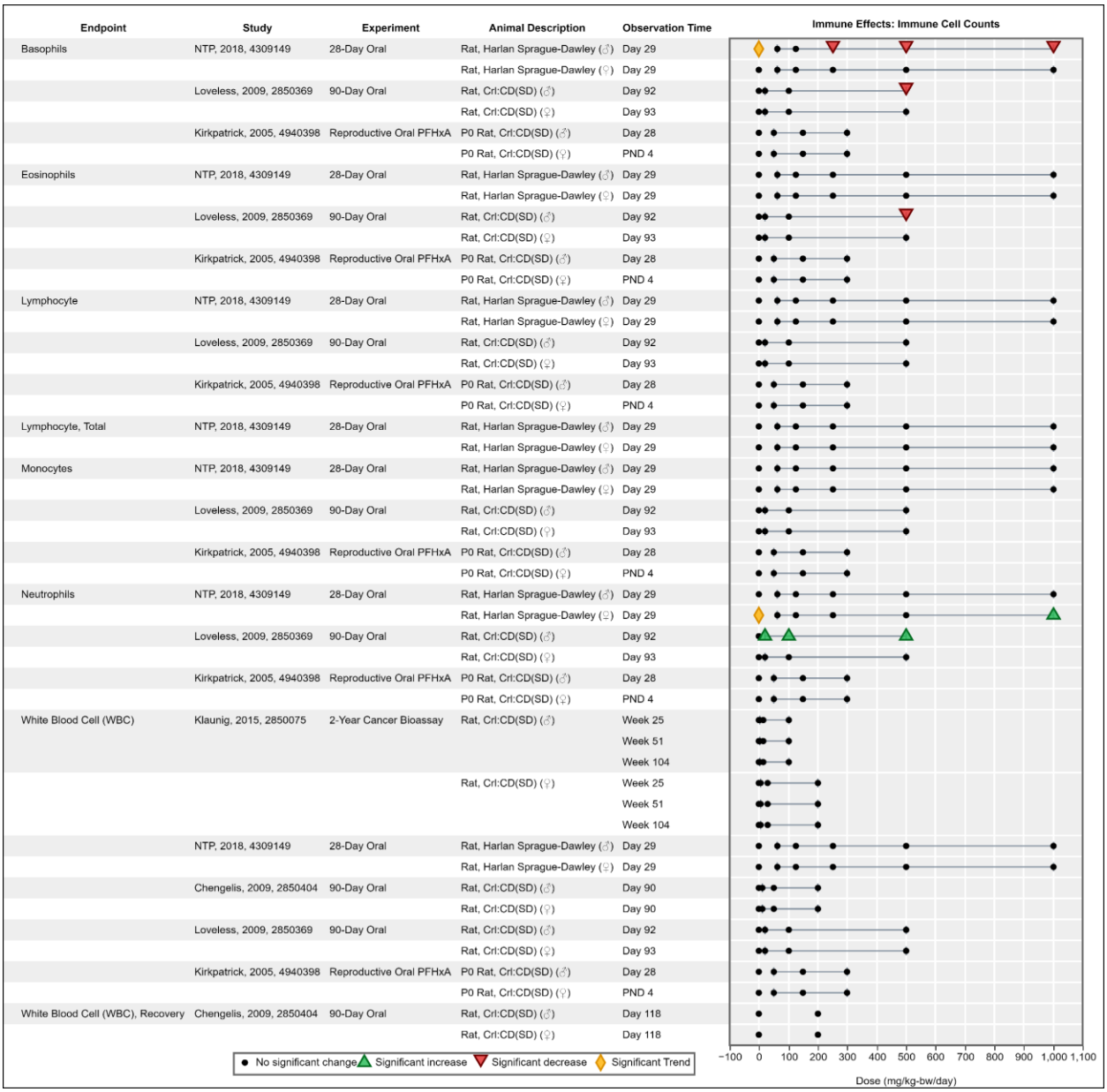


Figure 3-27. Immune cell counts in rats exposed by gavage to PFHxA or PFHxA sodium salt (HAWC: PFHxA – Animal Toxicity Immune Effects link).

Evidence Integration

The human evidence was limited to one *medium* confidence study that showed no clear association between PFHxA exposure and immune-related health outcomes, although the authors did observe a nonsignificant trend toward an association with asthma diagnosis in the previous year. Based on these results, there is *indeterminate* human evidence of immune effects.

Except for changes in thymus weight, the available animal toxicological evidence did not show a clear pattern of effect across studies. Specifically, three studies reported treatment-related changes in thymus and spleen weights in rats, but the direction of effect on spleen weights was not

consistent across studies. Extramedullary hematopoiesis was the only histopathological finding of note, but this is interpreted as possibly secondary to the effects on red blood cells rather than an immune-specific effect and is discussed in that context in Section 3.2.4. Increases in neutrophils and decreases in basophils showed a consistent direction of effect across two studies (of the five available). Eosinophils also were decreased, but only in males in a single study. No other treatment-related changes were observed for immune cell counts (i.e., specific cell populations or total white blood cells), and discerning the biological significance of this pattern is difficult in isolation.

The evidence supporting the potential immunotoxicity to humans is limited by several factors, including the lack of consistency across studies for several of the affected outcomes. Furthermore, the evaluated outcomes are limited to changes in the structural components of the immune system, which are less predictive indicators of immunotoxicity (IPCS, 2012). Notably, there is evidence indicating that other PFAS, including PFOS and PFOA, may affect immune system function through suppression of antibody response and induction of hypersensitivity (Dewitt et al., 2019). Additional studies, particularly those that evaluate changes in immune function would be beneficial for understanding the potential for adverse effects of PFHxA exposure on the immune system. Based on these results, there is *indeterminate* animal evidence of immune effects.

Overall, the currently available evidence is ***inadequate*** to determine whether PFHxA exposure might cause immune system effects in humans (see Table 3-35).

Table 3-35. Evidence profile table for immune effects

Evidence stream summary and interpretation					Evidence integration summary judgment
Evidence from studies of exposed humans					⊙⊙⊙ ***Evidence inadequate*** *Primary basis:* Evidence is *low* confidence or limited *Human relevance:* • N/A (*indeterminate* animal evidence) *Cross-stream coherence:* N/A (human evidence *indeterminate*) *Susceptible populations and lifestages:* • No evidence to inform
Studies and confidence	**Factors that increase certainty**	**Factors that decrease certainty**	**Summary and key findings**	**Evidence stream judgment**	
Asthma 1 *medium* confidence study	• No factors noted	• Imprecision • *Lack of coherence* – no associations with other measures of pulmonary function	• Nonsignificant association with asthma diagnosis, but other asthma-related outcomes were not affected.	⊙⊙⊙ ***Indeterminate***	
Evidence from animal studies					
Studies and confidence	**Factors that increase certainty**	**Factors that decrease certainty**	**Summary and key findings**	**Evidence stream judgment**	
Histopathology 4 *high* and *medium* confidence studies in adult rats: • 28-d • 90-d • 2-yr • Reproductive 1 *low* confidence study in rats: • 90-d	• *High* and *medium* confidence studies • *Consistency* across studies for extramedullary hematopoiesis	• No factors noted	• Increased splenic extramedullary hematopoiesis was observed male and female rats at 500 mg/kg-d; coincident with minimal erythroid hyperplasia of the bone marrow • Several immune-related histopathology findings reported in male and female rats at 300/450 mg/kg-d, but only in animals with unscheduled deaths	⊙⊙⊙ ***Indeterminate*** Some evidence of immune system but limited by unexplained inconsistency, lack of coherence, and potential for non-immune related causes [see Section 3.2.4 for additional discussion]. Available evidence was consisted of observational outcomes that are less predictive of immune system	

Evidence stream summary and interpretation					Evidence integration summary judgment
Immune Cell Counts 5 high and *medium* confidence studies in rats: • 28-d • 90-d (2 studies) • 2-yr • Reproductive	• *High* and *medium* confidence studies • *Consistency*–studies for neutrophils and basophils	• *Lack of coherence* with other immune markers	• Decreased basophil counts and increased neutrophil cell counts at ≥20 mg/kg-d	toxicity and in one study were only observed in that died or were removed from the study due to overt toxicity.	
Organ Weight 4 *high* and *medium* confidence studies in rats: • 28-d • 90-d (2 studies) • Reproductive	• *High* and *medium* confidence studies	• *Unexplained inconsistency* across studies for spleen weights	• Thymus weights decreased at 500 mg/kg-d in short-term, subchronic, and reproductive studies • Changes in spleen weight were inconsistent in the direction of effect across studies		
Mechanistic evidence and supplemental information					
Biological events of pathways	**Biological events of pathways**	**Biological events of pathways**		**Biological events of pathways**	
• No studies Identified					

3.2.9. Nervous System Effects

Human

No studies were identified that evaluated the effects of PFHxA on the nervous system in humans.

Animal

Four short-term (28-day), subchronic, and chronic animal studies evaluated the effects of PFHxA or PFHxA sodium salt in rats. Most outcome-specific study ratings were *high* or *medium* confidence. One study was rated *low* confidence for histopathology due to concerns about observational bias, endpoint sensitivity and specificity, and data presentation (Chengelis et al., 2009b). A summary of the studies and the interpretations of confidence in the results for the different outcomes based on the individual study evaluations is presented in Table 3-36, and details are available by clicking the HAWC link.

Table 3-36. Nervous system endpoints for PFHxA and associated confidence scores from repeated-dose animal toxicity studies

Author (year)	Species, strain (sex)	Exposure design	Exposure route and dose range	Brain weight	Histopathology	Behavior
NTP (2018)	Rat, Harlan Sprague-Dawley (male and female)	Short term (28 d)	Gavage[a] Male and female: 0, 62.5, 125, 250, 500, 1,000 mg/kg-d	++	++	NM
Chengelis et al. (2009b)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	Subchronic (90 d)	Gavage[a] Male and female: 0, 10, 50, 200 mg/kg-d	++	–	+
Loveless et al. (2009)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	Subchronic (90 d)	Gavage[b] Male and female: 0, 20, 100, 500 mg/kg-d	++	++	++
Klaunig et al. (2015)	Rat, Crl:CD(SD) Sprague-Dawley (male and female)	2-yr cancer bioassay	Gavage[a] Male: 0, 2.5, 15, 100 mg/kg-d Female: 0, 5, 30, 200 mg/kg-d	NM	++	++

++ Outcome rating of *high* confidence; + outcome rating of *medium* confidence; – outcome rating of *low* confidence; NM, outcome not measured.

[a,b]Study evaluation for animal toxicological nervous system endpoints reported from studies with male and female rats receiving PFHxA[a] or PFHxA sodium salt[b] by gavage. Study evaluation details for all outcomes are available by clicking the HAWC link.

Brain Weight

Three studies evaluated effects of PFHxA or PFHxA sodium salt on the nervous system in animals (NTP, 2018; Chengelis et al., 2009b; Loveless et al., 2009). Two studies reported increases in relative but not absolute brain weights after exposure to PFHxA or PFHxA sodium salt for 28 or 90 days, respectively (Chengelis et al., 2009b; Loveless et al., 2009). These effects were observed at the highest dose tested (200 or 500 mg/kg-day) and affected only males in one study (Loveless et al., 2009) and only females in the other (Chengelis et al., 2009b). Notably, relative weights are not considered appropriate for brain weight measurements because this measure is not typically affected by fluctuations in body weight (U.S. EPA, 1998); therefore, absolute brain weights are preferred.

Other Nervous System Effects

No treatment-related effects were observed on other nervous system outcomes, including behavior (i.e., open field locomotor activity, functional observational battery) and histopathology (NTP, 2018; Klaunig et al., 2015; Chengelis et al., 2009b; Loveless et al., 2009).

Mechanistic Evidence and Supplemental Information

Two studies evaluated effects of PFHxA exposure on neurodevelopment in zebrafish using wildtype (Guo et al., 2021; Gaballah et al., 2020) and transgenic [Tg (HuC-GFP)] (Guo et al., 2021) strains exposed during embryonic and early larval development. Both reported effects on larval swimming behavior, with larvae showing an increase in swimming activity at low to moderate doses but no effect at the higher doses. Guo et al. (2021) evaluated several other nervous system related outcomes in wildtype larvae, including acetylcholinesterase activity, and neurotransmitter levels. Acetylcholinesterase activity was statistically significantly reduced at all dose levels. Treatment-related effects on neurotransmitter levels was largely limited to the high exposure group (12 mg/L), with dopamine, DOPAC, and GABA levels showing 92% to 174% increases relative to controls. Acetylcholine was slightly reduced in the 2.4 mg/L exposure group and there was no effect on serotonin. A transgenic strain, Tg (HuC-GFP), that expresses GFP in neuronal cells was used to evaluate effects on neurodevelopment and proliferation in the central nervous system in vivo. Larvae in the high exposure group showed a 48% decreased in GFP expression, but there were no effects in the lower exposure groups.

Two studies evaluated effects on expression of genes and proteins related to neurodevelopment in primary avian neuronal cells (Vongphachan et al., 2011) and larval zebrafish (Guo et al., 2021). Only one gene was evaluated in both studies. *mbp*, which is essential for myelination of nerves, showed increased expression in both models, although this was only observed at the lowest tested concentration in the zebrafish study. Guo et al. (2021) examined 17 additional genes that are involved in various aspects of nervous system function and development in zebrafish larvae exposed up to 120 hours post fertilization. The results are summarized in in Table 3-37. Statistically significant changes in gene expression were reported for every gene.

Although some genes showed increased expression (*mbp, dat, bdnf, nr4a2b*), most were decreased. Protein expression data from the same study provide some support for these results. α1-tubulin, elavl3, and gap43 protein levels were reduced in animals exposed to med or high concentrations showing a similar pattern and direction of effect as the gene expression results. Vongphachan et al. (2011) also measured effects on *rc3*, which is important for Ca2+ signal transduction and may play a role in long term potentiation but found no treatment-related effects.

Table 3-37. Changes in expression of genes related to neurodevelopment in early life stage zebrafish

Gene	Function/Role	0.48 mg/L PFHxA	2.4 mg/L PFHxA	12 mg/L PFHxA
chrna7	Acetylcholine receptor	–	–	↓
ache	Acetylcholinesterase	–	↓	↓
mbp	Myelination	↑	–	–
α1-tubulin	Axon/dendrite growth	–	–	↓
shha	Axon growth	–	↓	↓
elavl3	Differentiation of ganglion, amacrine cells	–	–	↓
gap43	Synaptogenesis; axon development/regeneration	–	↓	↓
syn2a	Synaptogenesis	↑	↑	↓
gfap	Astroglia development	–	–	↓
th2	Serotoninergic neuron marker	↓	↓	–
htr1ab	Serotonin receptor	↓	↓	↓
htr1b	Serotonin receptor	↓	↓	↓
htr2a	Serotonin receptor	↓	↓	↓
htr1aa	Serotonin receptor	↓	↓	↓
htr5a	Serotonin receptor	–	–	↓
dat	Dopamine reuptake	–	–	↑
bdnf	Neuronal differentiation	↑	↑	–
nr4a2b	Development of dopaminergic neurons	↑	↑	↑

–, no statistically significant change relative to control; ↓, statistically significant decrease relative to control; ↑, statistically significant increase relative to control.

Evidence Integration

No human studies were identified to inform the potential nervous system effects of PFHxA or PFHxA salts; therefore, there is *indeterminate* human evidence of nervous system effects.

In animals, the only available evidence to support an effect of PFHxA or PFHxA salts the nervous system stems from increase in relative brain weights, which is not considered a reliable

measure of neurotoxicity (U.S. EPA, 1998). No treatment-related effects were reported for other nervous system outcomes.

Although the available animal toxicity data are largely null and derived from low risk of bias studies, some uncertainties and data gaps remain. The results are limited to a small number of studies in adult animals, and the evidence base is lacking studies that could inform potential for nervous system effects when exposure occurs during development. This lifestage is a known critical window of sensitivity for nervous system effects (U.S. EPA, 1998) and has been identified as a research area of potential concern for other PFAS known to affect thyroid function. Two studies provide limited mechanistic evidence and supporting information suggesting the potential for neurodevelopmental effects of PFHxA. Based on these results, there is *indeterminate* animal evidence of nervous system effects and additional studies would be necessary to draw a more definitive judgment.

Overall, the currently available ***evidence is inadequate*** to assess whether PFHxA might cause nervous system effects in humans (see Table 3-38).

Table 3-38. Evidence profile table for nervous system effects

<table>
<tr><th colspan="5">Evidence stream summary and interpretation</th><th>Evidence integration summary judgment</th></tr>
<tr><td colspan="5">Evidence from studies of exposed humans</td><td rowspan="11">⊙⊙⊙
Evidence inadequate
Primary Basis:
No evidence in humans and animal evidence is largely null or lacking biological relevance.

Human relevance:
N/A (indeterminate animal evidence)

• Cross stream coherence:
N/A (human evidence indeterminate).

Susceptible populations and lifestages:
No evidence to inform.

Other inferences:
Some mechanistic data and supplemental information were identified that provide limited support for potential PFHxA mediated nervous system effects. These were limited to neurodevelopmental models for which there are no available animal data and</td></tr>
<tr><td>Studies and confidence</td><td>Factors that increase certainty</td><td>Factors that decrease certainty</td><td>Summary and key findings</td><td>Strength of evidence</td></tr>
<tr><td colspan="4">• No studies identified</td><td>⊙⊙⊙
Indeterminate</td></tr>
<tr><td colspan="5">Evidence from animal studies</td></tr>
<tr><td>Studies and confidence</td><td>Factors that increase certainty</td><td>Factors that decrease certainty</td><td>Summary and key findings</td><td>Strength of evidence summary</td></tr>
<tr><td>Brain Weight
3 high confidence studies in adult rats:
• 28-d
• 90-d (2 studies)</td><td>• High confidence studies</td><td>• No factors noted</td><td>• Increased relative brain weights (preferred metric) in animals at ≥200 mg/kg-d; absolute brain weight unaffected</td><td rowspan="2">⊙⊙⊙
Indeterminate

Evidence is largely null. The only evidence of nervous system effects was relative brain weight increases, which is not considered to be appropriate for evaluating nervous system toxicity.</td></tr>
<tr><td>Histopathology
3 high confidence studies in adult rats:
• 28-d
• 90-d
• 2-yr
1 low confidence study in adult rats:</td><td>• High confidence studies</td><td>• No factors noted</td><td>• No treatment-related effects reported</td></tr>
</table>

Evidence stream summary and interpretation					Evidence integration summary judgment
• 90-d					there are insufficient to inform a potential MOA.
Behavior 2 *high* confidence studies in adult rats: • 90-d • 2-yr 1 *medium* confidence study in adult rats: • 90-d	• *High* and *medium* confidence studies	• No factors noted	• No treatment-related effects reported		
Mechanistic evidence and supplemental information					
Biological events or pathways	**Key findings, interpretation, and limitations**			**Evidence stream summary**	
Behavior	*Key findings and interpretation:* • Alterations in larval swimming activity observed following early life exposure in zebrafish • Increased swimming activity at low to moderate exposure levels in two studies • Statistically significant effects seen mostly at lowest exposure level Limitations: • Small evidence base with effects not showing a clear dose response			Some support for potential effects of PFHxA on neurodevelopment, but these are largely limited to a single study in early life stage zebrafish.	
Neurodevelopment	*Key findings and interpretation*: • Neuron-specific GPF expression statistically significantly reduced in vivo in larval zebrafish, suggesting treatment-related effects on neuro proliferation • Effects only observed in high exposure group *Limitations:*				

Evidence stream summary and interpretation			Evidence integration summary judgment
	• Small evidence base		
Acetylcholinesterase Activity	*Key findings and interpretation:* • Decreased acetylcholinesterase activity observed in larval zebrafish at in all dose groups *Limitations:* • Small evidence base		
Neurotransmitter Levels	*Key findings and interpretation:* • Altered neurotransmitter levels in whole body zebrafish larvae observed at all dose levels *Limitations:* • Small evidence base		
Gene and Protein Expression	*Key findings and interpretation:* • Changes observed in 18 genes and 3 proteins related to nervous system function and development in larval zebrafish • Decreases in both gene and protein expression for *a1-tubulin, elavl3, and gap43* in zebrafish • Two studies in zebrafish and avian neuronal cells found decreases in gene expression of *mbp*, but no clear dose response in zebrafish (low exposure group only) *Limitations:* • Small evidence base		

3.3. CARCINOGENICITY

3.3.1. Cancer

Human Studies

One *low* confidence case-control study of breast cancer and PFHxA exposure is available in humans (Velarde et al., 2022). This study reported a positive association in the fourth quartile (OR = 2.66, 95% CI: 0.95-7.66 vs. Q1), but the association was non-monotonic across quartiles (inverse associations were reported in the second and third quartiles) and there were serious concerns for potential selection bias and exposure misclassification (due to lack of temporality in exposure measurement), so there is considerable uncertainty in this finding.

Animal Studies

A *high* confidence cancer bioassay conducted in rats evaluated neoplastic and non-neoplastic lesions in the lungs, kidney, stomach, and liver of male rats dosed with 0, 2.5, 15, or 100 mg/kg-day and in female rats dosed with 0, 5, 30, or 200 mg/kg-day (Klaunig et al., 2015). Findings for nonneoplastic and neoplastic lesions were reported as null and are summarized in HAWC and in PFHxA Tableau.

Genotoxicity

Genotoxic, mutagenic, and clastogenic effects of PFHxA have been tested in several mammalian and prokaryotic cell systems in vitro (see Table 3-39) (Eriksen et al., 2010; Nobels et al., 2010; Loveless et al., 2009). Sodium perfluorohexanoate (NaPFHx) was negative for mutagenicity in *Escherichia coli* strain WP2*uvr*A and *Salmonella typhimurium* strains TA98, TA100, TA1535, and TA1537 in both the presence and absence of exogenous S9 metabolic activation (Loveless et al., 2009). Nobels et al. (2010) examined the ability of PFHxA to induce the expression of 14 prokaryotic stress response genes after exposure of the *E. coli* K-12 derivative SF_1 to 0.0156–1 mM PFHxA. The results of this study demonstrated that PFHxA did not significantly induce the expression of regulatory elements critical for the prokaryotic gene expression response to oxidative stress (*katG*, *zwf*, *soi28*, and *nfo*), membrane damage (*micF* and *osmY*), general cell lesions (*uspA* and *clpB*), heavy metal stress (*merR*), and DNA damage (*nfo*, *recA*, *umuDC*, *ada*, *sfiA*, and *dinD*). In mammalian cells in vitro, PFHxA did not generate reactive oxygen species (ROS) or oxidative deoxyribonucleic acid damage in the human hepatoma cell line, HepG2 (Eriksen et al., 2010). Lastly, NaPFHx failed to induce chromosomal aberrations in human peripheral blood lymphocytes in the presence and absence of exogenous metabolic activation, suggesting a lack of clastogenic activity (Loveless et al., 2009).

Evidence Integration

A single *low* confidence human study reported a positive association between PFHxA exposure measures and breast cancer; however, this association was limited to the fourth quartile with inverse associations reported for the second and third quartiles (Velarde et al., 2022). In animals, one study (Klaunig et al., 2015) evaluated the potential carcinogenicity of oral PFHxA exposure via histological evaluation of the lung, kidney, stomach, and liver of male rats, and did not observe significant treatment-related effects, and the few studies examining markers of potential genotoxicity were largely null. No studies of potential carcinogenicity in via other exposure routes were identified. Given the sparse and notably uncertain evidence base, and in accordance with the *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005) EPA concluded there is ***inadequate information to assess carcinogenic potential*** for PFHxA for any route of exposure.

Table 3-39. Summary of PFHxA genotoxicity studies

PFHxA genotoxicity						
			Results[a]			
Endpoint	Test system	Doses/ concentrations tested	Without exogenous activation	With exogenous activation	Comments	References
ROS production	HepG2 (human hepatoma cell line)	0.4, 4, 40, 200, 400, 1,000, 2,000 µM	–	NA	Intracellular reactive oxygen species (ROS) production was measured using 2',7'-dichlorofluorescein diacetate. ROS production was measured every 15 min for 3 hr. No clear concentration-response relationship was observed for PFHxA, whereas exposure to H_2O_2 (positive control) generated ROS in a concentration dependent manner.	Eriksen et al. (2010)
DNA damage	HepG2 (human hepatoma cell line)	100, 400 µM	–	NA	Comet assay to detect the formation of DNA strand breaks (including alkali-labile sites) and formamidopyrimidine-DNA-glycosylase sensitive sites after 24-hr exposure. Cytotoxicity was monitored by measuring lactate dehydrogenase (LDH) activity to ensure observed DNA damage was not secondary to cytotoxicity.	Eriksen et al. (2010)
Cell stress-dependent gene expression	*Escherichia coli* K-12 derivative SF_1	0.0156, 0.03125, 0.0625, 0.125, 0.25, 0.5, 1 mM	–	NA	Promoters of 14 prokaryotic DNA-damage responsive genes were fused to lacZ cassettes and expressed in *E. coli*. Activation of gene expression was measured after 90 min of exposure by β-galactosidase reduction capacity and spectrophotometrically at 420 nm. Genes involved in prokaryotic DNA damage and repair (*umuDC* and *ada*) were upregulated at approximately ≥1.4-fold but did not reach statistical significance at any dose. Study authors did not provide complete data for analysis.	Nobels et al. (2010)
Mutation (Ames assay)	*Salmonella typhimurium* strains TA98, TA100, TA1535, and TA1537	333, 667, 1,000, 3,333, 5,000 µg/plate sodium perfluorohexanoate (NaPFHx)	–	–	Assay performed according to OECD Guideline 471. No positive mutagenic responses were observed at any dose level or with any tester strain in the presence or absence of S9 metabolic activation.	Loveless et al. (2009)

PFHxA genotoxicity						
			Results[a]			
Endpoint	Test system	Doses/ concentrations tested	Without exogenous activation	With exogenous activation	Comments	References
Mutation	*E. coli* WP2uvrA	333, 667, 1,000, 3,333, 5,000 µg/plate sodium perfluorohexanoate (NaPFHx)	–	–	Assay performed according to OECD Guideline 471. No positive mutagenic responses were observed at any dose level or with any tester strain in the presence or absence of S9 metabolic activation.	Loveless et al. (2009)
Chromosomal aberration	Human peripheral blood lymphocytes	4h (nonactivated): 2,000, 3,000, 3,860 µg/mL sodium perfluorohexanoate (NaPFHx) 4 hr (activated) and 22 hr (nonactivated): 250, 500, 1,000 µg/mL sodium perfluorohexanoate (NaPFHx)	–	–	Assay performed according to OECD Guideline 473. Percentage of cells with structural or numerical aberrations was not significantly increased above that of the vehicle control at any concentration. Aroclor-induced rat liver S9 was used for exogenous metabolic activation. Mitomycin C and cyclophosphamide were used as positive controls. Substantial toxicity (defined as a reduction in the mitotic index of >50% in the NaPFHx treated cell culture as compared to vehicle control) was observed in all test conditions.	Loveless et al. (2009)
Cell Proliferation and neoplastic transformation	Human breast epithelial cells (MCF-10A)	500 pM, 1 nM, 10 nM, 100 nM, 500 nM, 1 µM, 10 µM, 100 µM, 500 µM	–	–	Cell proliferation was inferred from cell viability (measured by the MTT assay) and cell counting after 72 hrs PFAS treatment. No alterations were identified for PFHxA at any of the concentrations tested. Neoplastic transformation was measured by transwell migration and invasion assays. No positives were identified for PFHxA compared to controls at any of the concentrations tested.	(Pierozan et al., 2022)

PFHxA genotoxicity						
			Results[a]			
Endpoint	Test system	Doses/ concentrations tested	Without exogenous activation	With exogenous activation	Comments	References
DNA damage in sperm	Human sperm cells obtained from nonsmoker healthy men (n = 3, 27–34 yr old)	100 µM, 300 µM, 1000 µM	–	–	DNA damage was analyzed using the alkaline comet assay. No significant DNA damage was identified in human sperm cells treated with PFHxA.	(Emerce and Çetin, 2018)

[a]– = negative; NA = not applicable.

4. SUMMARY OF HAZARD IDENTIFICATION CONCLUSIONS

4.1. SUMMARY OF CONCLUSIONS FOR NONCANCER HEALTH EFFECTS

For all noncancer health effects, limited or no human epidemiological evidence was available. Therefore, conclusions were based primarily on animal toxicological studies. The animal evidence base consists of a short-term daily exposure to 0, 62.5, 125, 250, 500, 1000 mg/kg PFHxA for 28 days (NTP, 2018), subchronic daily exposure to 0, 10, 50, 200 mg/kg PFHxA (Chengelis et al., 2009b), subchronic daily exposure to 0, 20, 100, 500 mg/kg PFHxA sodium salt (Loveless et al., 2009), and chronic daily exposure for two years to 0, 5, 30, 200 mg/kg or 0, 2.5, 15, 100 mg/kg PFHxA female or male respectively (Klaunig et al., 2015). Two developmental, gestational exposure, studies (Iwai and Hoberman, 2014; Loveless et al., 2009) and a one-generation reproductive study (Loveless et al., 2009) with maternal oral doses between 7–500 mg/kg-day also were also available. The outcome-specific ratings for these studies were generally *high* confidence.

As described in detail in Section 3, the available ***evidence indicates*** that PFHxA exposure is likely to cause hepatic (Section 3.2.1), developmental (Section 3.2.2), hematopoietic (Section 3.2.4), and endocrine effects (Section 3.2.5) in humans, given sufficient exposure conditions. As previously noted, the "sufficient exposure conditions" are more fully evaluated and defined for the identified health effects through dose-response analysis in Section 5.

The evidence for PFHxA-mediated adverse hepatic effects was based primarily on a set of consistent and coherent findings in animal studies, including hepatocellular hypertrophy and increased relative liver weight, generally at ≥100 mg/kg-day PFHxA. Most of the hepatic effects observed in rodents changed in a consistent direction between sexes and across studies, however the magnitude of change was often greater in males compared with females possibly due to faster PFHxA clearance in females compared with males (see Section 3.1.4). Hepatic hypertrophy persisted after recovery in the subchronic studies and was coherent with findings of increased peroxisomal beta oxidation in animals. Hepatic hypertrophy was not observed after chronic exposure, possibly because of the tested dose levels (e.g., male rats in the chronic study were maximally exposed to PFHxA doses 2- to 5-fold below the maximal doses in the two subchronic studies) or other study design differences. Interestingly, necrosis was observed in the female high dose group (200 mg/kg-day) in the chronic study, suggesting that with longer term exposure, hepatocellular hypertrophy could progress to more severe outcomes such as necrosis. These effects in rodents were evaluated together along with supplemental evidence, including mechanistic and pharmacokinetics information, to consider whether the rodent effects represent adverse or adaptive changes to PFHxA exposure and their human relevance. Regarding human relevance, the

hepatocellular hypertrophy findings were coherent with mechanistic findings on the activation of target genes and transcription factors, including PPARα, in human and rat cell systems treated with PFHxA. While the role of PPARα was not specifically challenged in rodent models exposed to PFHxA, evidence that PFHxA activates both human and rodent PPARα at comparable concentrations, alongside evidence from structurally similar PFAS, indicates that PPARα dependent and independent hepatic effects are both likely contributing to the hepatic effects of PFHxA and that these effects are relevant to humans. Regarding adversity, the histologic changes (increased hypertrophy and necrosis) were coherent with clinical pathology findings (including increased ALT and decreased serum globulins) and collectively considered in the context of the Hall criteria (see Section 3.2.1) as adverse.

The data from the animal toxicological studies that support identifying developmental effects as a potential human hazard included effects from three studies that reported consistent, dose-responsive, and concerning effects of PFHxA exposure on offspring body weights and mortality. Delayed eye opening was also reported. The observed developmental effects following PFHxA exposure are similar to other PFAS, including PFBS (U.S. EPA, 2021b) and PFBA (U.S. EPA, 2022b), providing additional support for these specific findings. Effects on offspring body weight were observed in two species (rats and mice) exposed to different PFHxA salts (sodium and ammonium) using different exposure scenarios, although effects on mortality were observed only in the mouse study. Low birth weight is associated with lasting adverse effects on health, with increased risk for disease and reductions in lifespan in humans (Thompson and Regnault, 2011; Kajantie et al., 2005; Barker, 2004) and delays in eye opening can impact normal development of the visual system (Espinosa and Stryker, 2012; Wiesel, 1982).

The primary support for hematopoietic effects included consistent decreases in red blood cells, hematocrit, and hemoglobin across study designs and exposure durations in male and female adult rats (NTP, 2018; Chengelis et al., 2009b; Loveless et al., 2009). These hematological findings correlate with increases in reticulocytes, an indicator of erythroid cell regeneration supported by pathological findings in the spleen and bone marrow (Loveless et al., 2009). The decreases in hemoglobin were consistent with the decreased mean corpuscular hemoglobin concentration observed in both sexes (NTP, 2018; Loveless et al., 2009). When combined, increased mean corpuscular hemoglobin concentration (MCHC), and mean corpuscular volume (MCV) are indicators of anemia. Several of the hematological findings were significant at the highest dose tested in the subchronic studies and returned to control levels after 30- or 90-day recovery periods (or both) (Chengelis et al., 2009b; Loveless et al., 2009). Findings from females in the chronic study (e.g., HGB, RBC, and reticulocytes) were significant at the highest administered dose (200 mg/kg-day), whereas no effects were observed in males that received half (100 mg/kg-day) the female dose. Together, the subchronic and chronic evidence from males and female rats suggest PFHxA-mediated hematopoietic effects are dependent on both dose and duration.

For endocrine effects, the evidence to support PFHxA mediated effects is largely based on a short-term study in rats showing a strong dose-dependent effect on thyroid hormones (decreased serum T4 and T3, with no significant changes in TSH) in males only (NTP, 2018). This pattern of thyroid hormone changes in rats is consistent with findings for other short-chain PFAS with more substantial evidence bases, namely PFBA and PFBS, increasing the certainty in this evidence. These rodent PFHxA data are also supported by similar findings in early lifestage zebrafish (Zhang et al., 2022). Effects on thyroid histopathology and thyroid weight were also investigated in several rat studies ranging from short-term to chronic exposure durations (NTP, 2018; Klaunig et al., 2015; Chengelis et al., 2009b; Loveless et al., 2009). Of these, only Loveless et al. (2009) found treatment related increases in thyroid follicular cell hypertrophy and organ weight. Additionally, some mechanistic evidence is available that suggests PFHxA may affect thyroid function by altering expression of thyroid-associated mRNA and proteins or binding with serum transport proteins or the nuclear receptor.

There ***evidence is inadequate*** to determine whether PFHxA has the potential to cause renal, male, and female reproductive, immune, and nervous system effects in humans. renal, male, and female reproductive, immune, and nervous system. There was ***insufficient evidence*** to determine whether PFHxA exposure has the potential to cause other health effects (e.g., ocular effects, cardiovascular, respiratory, gastrointestinal). Other potential health outcomes have not been evaluated in the context of PFHxA exposure. Thus, important data gaps for PFHxA exists given the associations observed for other PFAS, such as PFBS, PFOA, PFOS, and GenX (U.S. EPA, 2021a; MDH, 2020, 2019, 2018; U.S. EPA, 2018b, 2016a, b). See Table 4-1 for a comparison of the noncancer hazard judgments drawn for PFHxA with the judgments in the final EPA assessments for PFBS, PFOA, PFOS, and GenX.

Table 4-1. Hazard conclusions across published EPA PFAS human health assessments

Health outcome	PFAS assessments[a,b,c]					
	PFHxA (this assessment)	PFBA U.S. EPA (2022b)	PFBS[d] U.S. EPA (2018b)	Gen X chemicals[d] U.S. EPA (2021a)	PFOA[d] U.S. EPA (2023b)	PFOS[d] U.S. EPA (2023a)
Endocrine/ Thyroid	+	+	+	ND	Human: + Animal: +/−	Human: +/− Animal: +/−
Hepatic/Liver	+	+	−	+	Human: + Animal: +	Human: − Animal: +
Developmental	+	+	+	+/−	Human: + Animal: +	Human: + Animal: +
Reproductive	−	−	−	+/−	Human: − Animal: +/−	ND
Immunotoxicity	−	−	−	+/−	Human: + Animal: +	Human: +/− Animal: +
Renal	−	−	+	+/−	Human: +/− Animal: +/−	ND
Hematopoietic/ Hematological	+	−	ND	+/−	ND	ND
Ocular	ND	−	ND	ND	ND	ND
Serum Lipids	ND	ND	−	ND	Human: + Animal: +	Human: +
Hyperglycemia	ND	ND	ND	ND	Human: − Animal: −	Animal: +/−
Nervous System	−	ND	ND	ND	Human: − Animal: −	Animal: +/−

Health outcome	PFAS assessments[a,b,c]					
	PFHxA (this assessment)	**PFBA U.S. EPA (2022b)**	**PFBS[d] U.S. EPA (2018b)**	**Gen X chemicals[d] U.S. EPA (2021a)**	**PFOA[d] U.S. EPA (2023b)**	**PFOS[d] U.S. EPA (2023a)**
Cardiovascular	ND	ND	–	ND	ND	ND
Cancer	–	–	–	+/–	+/–	+/–

[a]Assessments used multiple approaches for summarizing their noncancer hazard conclusion scales; for comparison purposes, the conclusions are presented as follows: '+' = evidence demonstrates or evidence indicates (e.g., PFHxA), or evidence supports (e.g., PFBS); '+/–' = suggestive evidence, '–' = inadequate evidence (e.g., PFHxA) or equivocal evidence (e.g., PFBS); '–/–' = sufficient evidence to conclude no hazard (no assessment drew this conclusion); ND = no data available for this outcome for this PFAS.

[b]The assessments all followed the EPA carcinogenicity guidelines (U.S. EPA, 2005) a similar presentation to that used to summarize the noncancer judgments is applied for the cancer hazard conclusions, as follows: '+' = carcinogenic to humans or likely to carcinogenic to humans;'+/–' = suggestive evidence of carcinogenic potential; '–' = inadequate information to assess carcinogenic potential; '–/–' = not likely to be carcinogenic to humans(no assessment drew this conclusion); ND = no carcinogenicity data available for this PFAS.

[c]The hazard conclusions for the various EPA PFAS assessments presented in this table were not considered during evidence integration and thus did not inform the evidence integration conclusions presented in the PFHxA assessment.

[d]The U.S. EPA PFOA (U.S. EPA, 2016b) and PFOS (U.S. EPA, 2016a) assessments did not use structured language to summarize the noncancer hazard conclusions. The presentation in this table was inferred from the hazard summaries found in the respective assessments; however, this is for comparison purposes only and should not be taken as representative of the conclusions from these assessments. Those interested in the specific noncancer hazard conclusions for PFOA and PFOS must consult the source assessments. Note that new assessments for PFOA and PFOS are currently being finalized to support a National Primary Drinking Water Regulation; note that hazard conclusions in these updated assessments will differ from those presented in this table as the new assessments use structured language to summarize the noncancer hazard conclusions. For access to the more recent draft assessment materials please follow this link.

4.2. CONCLUSIONS REGARDING SUSCEPTIBLE POPULATIONS AND LIFESTAGES

No human studies were available to inform the potential for PFHxA exposure to affect sensitive subpopulations or lifestages.

In adult rats exposed to PFHxA for 28 days to 2 years, toxicological findings were either consistently observed at lower dose levels in males than females or the findings were observed only in males (except for necrosis in the chronic study). The reason for this sex dependence is possibly due to sex-dependent PFHxA elimination caused by sex-specific differences in the expression (mRNA and protein) of the renal organic anion transporting polypeptide (Oatp) 1a1 (Kudo et al., 2001) as discussed in Section 3.1.4. Currently, whether this sex-specific difference might also exist in humans is unclear.

Additionally, given the effects seen in the developing organism (i.e., perinatal mortality, reduced body weights, and delays in time to eye opening), the prenatal and early postnatal window represents a potentially sensitive lifestage for PFHxA exposure.

4.3. SUMMARY OF CONCLUSIONS FOR CARCINOGENICITY

The evidence is inadequate to make a judgment on whether PFHxA exposure might affect the development of any specific cancers (see Section 3.3). Given this general lack of evidence and consistent with EPA guidelines (U.S. EPA, 2005) instruction to apply a standard descriptor as part of the hazard narrative and to express a conclusion regarding the weight of evidence for the carcinogenic hazard potential, a descriptor of ***inadequate information to assess carcinogenic potential*** is applied for PFHxA; this descriptor is applicable across all exposure routes.

5. DERIVATION OF TOXICITY VALUES

5.1. HEALTH EFFECT CATEGORIES CONSIDERED (CANCER AND NONCANCER)

Multiple noncancer health effects were examined following oral PFHxA exposures. The evidence integration judgments are based primarily on five animal toxicological studies (NTP, 2018; Klaunig et al., 2015; Iwai and Hoberman, 2014; Chengelis et al., 2009b; Loveless et al., 2009). These studies were generally rated *high* confidence in outcome-specific study evaluations. Based on these studies, it was determined that the ***evidence indicates*** PFHxA likely causes hepatic, developmental, hematopoietic, and endocrine effects in humans given sufficient exposure conditions. These health effects were considered for derivation of toxicity values. The dose levels associated with these hazards are further characterized in Section 5.2.1.

For all other health effects (i.e., renal, male, and female reproductive, immune, and nervous system), the ***evidence is inadequate*** to assess potential health effects, thus these were not considered for derivation of toxicity values.

No studies of inhalation exposure were identified, thus an RfC was not estimated (see Section 5.2.2). Similarly, the evidence base related to potential carcinogenicity was determined to contain "***inadequate information to assess carcinogenic potential***"; therefore, no cancer toxicity values were estimated for any exposure route (see Section 5.3).

5.2. NONCANCER TOXICITY VALUES

A reference dose (RfD) is the daily oral exposure to the human population (including sensitive subpopulations) that is likely without appreciable risk of deleterious effects during a lifetime. In addition to developing an RfD designed to protect against lifetime exposure, a less-than-lifetime toxicity value (referred to as a "subchronic RfD") is estimated. These subchronic toxicity values are presented as they might be useful for certain decision purposes (e.g., site-specific risk assessments with less-than-lifetime exposures). Both RfD and subchronic RfD derivations include organ/system-specific RfDs (osRfDs) associated with each health effect considered for point of departure (POD) derivation. Subsequent decisions related to dosimetric extrapolation, application of uncertainty factors, and confidence in toxicity values are discussed below.

As noted above, a reference concentration (RfC) or subchronic RfC could not be developed.

5.2.1. Oral Reference Dose (RfD) Derivation

Study Selection

The identified hazards relating to developmental, hepatic, hematopoietic, and endocrine effects were based on five *high* confidence animal studies that were considered for use in deriving an oral reference dose (RfD). The developmental studies Loveless et al. (2009) or Iwai and Hoberman (2014) were selected to support RfD derivation for developmental effects, given that endpoints observed from exposures during early lifestages are known to be sensitive for developmental effects (compared with exposures at later lifestages) and thus more appropriate for estimating potential effects of lifetime exposure. Both studies used rats or mice as the laboratory animal species and used vehicle-exposed controls. Animals were exposed to reagent-grade Na^+PFHxA (reported as 100% pure) or NH_4^+PFHxA (reported as 93.4% pure; impurities not reported) via a human-relevant route (oral administration via gavage) and for a duration of exposure (GD 6–18) encompassing critical windows relevant to the developmental effects of interest.

In addition to the developmental studies, subchronic (Chengelis et al., 2009b; Loveless et al., 2009) and chronic (Klaunig et al., 2015) studies in rats were selected to support RfD derivation for hepatic and hematopoietic outcomes of PFHxA exposure. Animals were exposed to Na^+PFHxA (reported as 100% pure) or PFHxA (reported as 98.1% pure) via oral administration and for a duration relevant for estimating potential effects of lifetime exposure (90 days or 2 years). For endocrine outcomes, endpoints were available from the same subchronic and chronic studies, however observations were null except for increased thyroid follicular cell hypertrophy from the 90-day study (Loveless et al., 2009) in males and females. However, the 90-day findings were only observed at the highest dose, were inconsistent with findings from the other 90-day and chronic study and lacked support for coherence from other findings. Also available in the PFHxA database is a short-term (i.e., 28-day) study of PFHxA exposure in rats (NTP, 2018) that was interpreted to provide critical information supporting information supporting thyroid hazard identification and included quantitative data useful for dose response analysis. Note that, as discussed below, when developing a lifetime reference value, chronic or subchronic studies (and studies of developmental exposure) are generally preferred over short-term or acute studies, so the (NTP, 2018) study was ultimately not considered for use in deriving candidate lifetime toxicity values.

For hepatic outcomes, a collection of adverse effects was found in rats, with PFHxA exposure resulting in increased liver weights (absolute and relative) in adult exposed animals (Chengelis et al., 2009b; Loveless et al., 2009), histopathological lesions (i.e., hepatocellular necrosis), and hypertrophy. Other hepatic findings (increased liver enzymes [i.e., ALT, AST, ALP]; increased peroxisomal beta oxidation; decreased total protein, albumin, and bilirubin; and mechanistic evidence indicating PPARα activation in humans and rodents at similar concentrations of PFHxA) supported the final evidence integration judgments for these endpoints and were critical for identifying adverse, human relevant endpoints for dose-response analysis (as discussed in

Section 3.2.1). For the purposes of dose-response modeling, hepatic hypertrophy was considered a more sensitive effect (compared with other hepatic findings such as relative liver weight). Hepatocellular hypertrophy was observed in both sexes by Loveless et al. (2009), whereas Chengelis et al. (2009b) observed hypertrophy in the livers of male mice, not females.

For hematopoietic outcomes, a collection of adverse effects including decreased HGB and decreased RBCs, which were coherent with other correlative changes in other red blood cell indicators as discussed in Section 3.2.4, were observed in PFHxA exposed rats. The observed HGB and RBC findings were considered the most sensitive and specific compared to other hematological findings (e.g., blood proteins, hematocrit) and both advanced for POD derivation as there was no reason to choose one endpoint over the other. Hemoglobin decreases were considered similar in sensitivity to decreases in red blood cell counts and, while hemoglobin is used as an indicator of disease of the blood (i.e., anemia) (WHO, 2022), there was no reason to advance one endpoint over the other. Hematopoietic effects were observed in both sexes in the subchronic studies, whereas the only significant observation from the chronic study was in females, likely due to females receiving the highest dose administered in the study (Klaunig et al., 2015). As described in Section 3.2.4, quantitative hematology measures in rats after 52 weeks are likely less reliable due to increasing naturally occurring diseases that decrease sensitivity; thus, no hematopoietic data after 51 weeks were considered for POD derivation.

For developmental outcomes adverse effects were observed that included decreased F_1 pup body weight and increased perinatal mortality after gestational exposure (Iwai and Hoberman, 2014; Loveless et al., 2009). Effects on F_1 pup body weight were strongest during the early postnatal period so these timepoints were prioritized. Perinatal mortality (still birth and postnatal deaths from PND 0–21) showed a clear dose-response across two experimental cohorts with overlapping dose ranges. Data were pooled for dose-response analysis. Eye opening delays were also observed but were not advanced for POD derivation because the PFHxA doses causing these effects were higher than those that lead to pup body weight deficits and perinatal mortality.

For endocrine outcomes, adverse effects related to the thyroid consisted of decreased free and total T4 and T3 levels observed in exposed rats in the short-term study (NTP, 2018), and increased thyroid follicular hypertrophy in a subchronic study (Loveless et al., 2009). Decreased thyroid hormone levels are judged relevant to human health, given the many similarities in the production, regulation, and functioning of thyroid hormones between rodents and humans. In addition, rodents are more sensitive to increases in thyroid follicular hypertrophy that was observed in one subchronic study compared with null findings from all other studies at similar and higher PFHxA dose levels (including in a chronic study). Thus, changes in thyroid hormone levels are considered more relevant for deriving human health toxicity values and increased thyroid hypertrophy was not considered further for RfD derivation. Total T4 assay measurements are more reliable than those provided by the assays available to measure free T4 due to the very small

quantity of unbound (i.e., 'free') T4 in circulation (Faix, 2013; Thienpont et al., 2013). For this reason, total, but not free, T4 was moved forward for POD and candidate value derivation.

A summary of endpoints considered for toxicity value derivation is presented in Table 5-1.

Table 5-1. Endpoints considered for dose-response modeling and derivation of points of departure

Endpoint	Study, endpoint confidence, exposure duration	Strain, species, sex	POD derivation[a]	Rationale for deriving POD
Hepatic				
Increased relative liver weight	Chengelis et al. (2009b) *High* confidence, Subchronic (90 d)	Crl:CD(SD) rat, both	No	Liver weights were considered a less sensitive and specific measure of toxicity than other available hepatic findings such as increased hepatocellular hypertrophy.
	Loveless et al. (2009) *High* confidence, Subchronic (90 d)	Crl:CD(SD) rat, both	No	
Increased hepatocellular hypertrophy	Chengelis et al. (2009b) *Low* confidence, Subchronic (90 d)	Crl:CD(SD) rat, both	No	Endpoint considered adverse based on collection of hepatic findings (see Section 3.2.1) and a biologically plausible precursor event of more severe outcomes such as necrosis. Although male-specific effects were observed in Chengelis et al. (2009b), the evaluation of this outcome in this study was considered low confidence and therefore not advanced. Both sexes were affected in Loveless et al. (2009) and advanced.
	Loveless et al. (2009) *High* confidence, Subchronic (90 d)	Crl:CD(SD) rat, both	Yes	
Increased hepatocellular necrosis	Klaunig et al. (2015) *High* confidence, Chronic (104 w)	Crl:CD(SD) rat, female	Yes	Although significant effects were only observed at the highest dose in females, and largely in animals that died an unscheduled death, necrosis was advanced for further consideration as it represents a severe and specific indication of hepatic toxicity.
Hematopoietic				
Decreased blood proteins (total protein and globulin)	Chengelis et al. (2009b) *High* confidence, Subchronic (90 d)	Crl:CD(SD) rat, both	No	Several hematologic endpoints are available and some (e.g., red blood cells and hemoglobin) are considered a more direct measure of hematopoietic effects than these markers.
	Loveless et al. (2009) *High* confidence, Subchronic (90 d)	Crl:CD(SD) rat, both	No	
	Klaunig et al. (2015) *High* confidence, Chronic (51 w)	Crl:CD(SD) rat, both	No	

Endpoint	Study, endpoint confidence, exposure duration	Strain, species, sex	POD derivation[a]	Rationale for deriving POD
Decreased hematocrit	Chengelis et al. (2009b) *High* confidence, Subchronic (90 d)	Crl:CD(SD) rat, both	No	Hematocrit is a measure of the percentage by volume of red blood cells in blood. The endpoint was considered a less direct measure of hematopoietic outcomes compared with other endpoints (e.g., red blood cells and hemoglobin) that were also available from the same studies.
	Loveless et al. (2009) *High* confidence, Subchronic (90 d)	Crl:CD(SD) rat, both	No	
	Klaunig et al. (2015) *High* confidence, Chronic (51 wk)	Crl:CD(SD) rat, female	No	
Decreased hemoglobin	Chengelis et al. (2009b) *High* confidence, Subchronic (90 d)	Crl:CD(SD) rat, both	Yes	The endpoint was considered a sensitive and specific adverse hematopoietic outcome (see Section 3.2.4). Male endpoints from the chronic study were mostly null and not advanced for POD derivation.
	Loveless et al. (2009) *High* confidence, Subchronic (90 d)	Crl:CD(SD) rat, both	Yes	
	Klaunig et al. (2015) *High* confidence, Chronic (51 wk)	Crl:CD(SD) rat, female	Yes	
	Klaunig et al. (2015) *High* confidence, Chronic (51 wk)	Crl:CD(SD) rat, male	No	
Decreased red blood cells	Chengelis et al. (2009b) *High* confidence, Subchronic (90 d)	Crl:CD(SD) rat, both	Yes	The endpoint was considered a biologically significant endpoints that is sensitive and specific for hematopoietic outcomes. There was no clear reason to advance one study over others.
	Loveless et al. (2009) *High* confidence, Subchronic (90 d)	Crl:CD(SD) rat, both	Yes	
	Klaunig et al. (2015) *High* confidence, Chronic, (51 wk)	Crl:CD(SD) rat, female	Yes	
Increased reticulocytes	Chengelis et al. (2009b) *High* confidence, Subchronic (90 d)	Crl:CD(SD) rat, female	No	Increases were considered to reflect a compensatory (secondary) response to decreased red blood cell parameters.
	Loveless et al. (2009) *High* confidence, Subchronic (90 d)	Crl:CD(SD) rat, both	No	
	Klaunig et al. (2015) *High* confidence, Chronic, (51 wk)	Crl:CD(SD) rat, both	No	
Developmental				

Endpoint	Study, endpoint confidence, exposure duration	Strain, species, sex	POD derivation[a]	Rationale for deriving POD
Decreased, postnatal (F_1) pup body weight	Loveless et al. (2009) *High* confidence, One-generation repro-ductive; measured on PND 0, 4, 7, 14, 21	Crl:CD(SD) rat, F_1	Yes	Effects on body weight were strongest during the early postnatal period so these timepoints were prioritized.
	Iwai and Hoberman (2014) *High* confidence, Develop-mental (GD 6–18); measured on PND 0, 4, 7, 14, 21	CD-1 mouse, F_1	Yes	
Decreased, F_1 fetal body weight	Loveless et al. (2009) *High* confidence, Develop-mental (GD 6–20); measured on GD 21	Crl:CD(SD) rat, F_1	No	Statistically nonsignificant 9% decrease only at the highest dose.
Increased, perinatal mortality	Iwai and Hoberman (2014) *High* confidence, Develop-mental (GD 6–18); measured on PND 0–21, including stillbirths	CD-1 mouse, F_1	Yes	Perinatal mortality (still birth and postnatal deaths from PND 0–21) showed a clear dose-response across two experimental cohorts with overlapping dose ranges. These data were pooled for dose-response analysis.
Delayed eye opening	Iwai and Hoberman (2014) *High* confidence, Develop-mental (GD 6–18); measured on PND 10–17	CD-1 mouse, F_1	No	Delays were observed at a dose that elicited body weight deficits and perinatal mortality.
Endocrine				
Decreased thyroxine (T4), total	NTP (2018) *High* confidence, Short term (28 d)	Harlan Sprague-Dawley rat, male	Yes	Males showed statistically significant decreases in free and total T4 and T3 at all doses. No effects were observed in females. Total T4 was prioritized due to a clear dose response pattern across the tested exposure range and increased assay reliability (vs. free T4).
	NTP (2018) *High* confidence, Short term (28 d)	Harlan Sprague-Dawley rat, female	No	
Decreased thyroxine, free (fT4)	NTP (2018) *High* confidence, Short term (28 d)	Harlan Sprague-Dawley rat, male	No	
	NTP (2018) *High* confidence, Short term (28 d)	Harlan Sprague-Dawley rat, female	No	

Endpoint	Study, endpoint confidence, exposure duration	Strain, species, sex	POD derivation[a]	Rationale for deriving POD
Decreased triiodothyronine (T3)	NTP (2018) *High* confidence, Short term (28 d)	Harlan Sprague-Dawley rat, male	No	
	NTP (2018) *High* confidence, Short term (28 d)	Harlan Sprague-Dawley rat, female	No	
Increased Thyroid epithelial cell hypertrophy	Loveless et al. (2009) *High* confidence, Subchronic (90 d)	Crl:CD(SD) rat, both	No	Increased incidence in thyroid epithelial hypertrophy only observed in one subchronic study at the high dose (Loveless et al., 2009).
	Chengelis et al. (2009b) *High* confidence, Subchronic (90 d)	Crl:CD(SD) rat, both	No	
	Klaunig et al. (2015) *High* confidence, Chronic (104 w)	Crl:CD(SD) rat, both	No	
	NTP (2018) *High* confidence, Short term (28 d)	Harlan Sprague-Dawley, both	No	
Increased thyroid weight	NTP (2018) *High* confidence, Short term (28 d)	Harlan Sprague-Dawley, both	No	Treatment-related effects were limited to a single study that only qualitatively reported increased thyroid weight in high dose group females.
	Loveless et al. (2009) *High* confidence, Subchronic (90 d)	Crl:CD(SD) rat, both	No	
	Chengelis et al. (2009b) *High* confidence, Subchronic (90 d)	Crl:CD(SD) rat, both	No	

[a]See text for rationale for inclusion/exclusion from point-of-departure derivation.

Estimation or Selection of Points of Departure (PODs)

The outcomes determined most appropriate for quantifying the identified noncancer hazards and advanced for dose-response analysis (see Table 5-1) were modeled using approaches consistent with EPA's *Benchmark Dose (BMD) Technical Guidance* document (U.S. EPA, 2012a). Specifically, the BMD and 95% lower confidence limit on the BMD (BMDL) were estimated using a benchmark response (BMR) to represent a minimal, biologically significant level of change. BMD modeling of continuous data was conducted using EPA's Benchmark Dose Software (BMDS, Version 3.2).

Ideally, the selected BMR is based on data that support the biological relevance of the outcome being evaluated; however, in some cases there is no clear scientific understanding to support a biologically based BMR. In these instances, the BMD guidance provides some BMRs that can be applied to the data. For data drawn from toxicological studies, a suggested BMR of 1 standard deviation (SD) from the control mean for continuous data or a BMR of 10% extra risk (ER) for dichotomous data can be used to estimate the BMD and BMDL. The selection of these BMRs, as indicated in Table 5-2, is based on BMD guidance stating that in the absence of information regarding the level of change considered biologically significant, these BMRs can be used (U.S. EPA, 2012a). For effects on offspring body weights, a BMR of 5% relative deviation (RD) from the control mean is used for continuous data to account for effects occurring in a sensitive lifestage (U.S. EPA, 2012a).

Table 5-2. Benchmark response levels selected for BMD modeling of PFHxA health outcomes

Endpoint	BMR	Rationale
Hepatic effects		
Hepatocellular hypertrophy	10% ER	For dichotomous hepatic data, a 10% ER is generally considered a minimally biologically significant response level (U.S. EPA, 2012a).
Hepatocellular necrosis	10% ER	For dichotomous hepatic data, a 10% ER is generally considered a minimally biologically significant response level (U.S. EPA, 2012a).
Developmental effects		
Postnatal (F_1) body weight	5% RD	A 5% RD in markers of growth/development in gestational studies (e.g., fetal weight) has generally been considered a minimally biologically significant response level and has been used as the BMR for benchmark dose modeling in prior IRIS assessments (U.S. EPA, 2012b, 2004, 2003).
Offspring mortality	1% ER	Although 5% ER is generally supported for developmental and reproductive outcomes (U.S. EPA, 2012a), a lower BMR of 1% ER was considered appropriate for modeling offspring mortality considering the severity of the frank effect.
Hematopoietic effects		
Red blood cells	1 SD	No biological information is readily available that allows for determining a minimally biological significant response for these outcomes. The BMD Technical Guidance (U.S. EPA, 2012a) recommends a BMR based on 1 SD in such a situation.
Hemoglobin		

Endpoint	BMR	Rationale
Endocrine effects		
Total T4	1 SD	Evidence to support identification of a minimally biologically significant response for thyroid hormones is lacking in adult animals. In developing animals, there is a wide range in the level of response in thyroid hormones associated with neurodevelopmental effects (10%–25% for serum T4) in human and rodent studies (Gilbert et al., 2016; Gilbert, 2011; Haddow et al., 1999). Given that the biological information is not sufficient to identify the BMR and the decreases in serum T4 (up to 73%) in male rats exceed these values, a 1 SD for continuous data was selected for this endpoint, consistent with EPAs Benchmark Dose Technical Guidance (U.S. EPA, 2012a).

An adequate fit is judged based on $\chi 2$ goodness-of-fit p-value ($p > 0.1$), magnitude of the scaled residuals in the vicinity of the BMR, and visual inspection of the model fit. In addition to these three criteria for judging adequacy of model fit, a determination is made as to whether the variance across dose groups is homogeneous. If a homogeneous variance model is deemed appropriate based on the statistical test provided by BMDS (i.e., Test 2), the final BMD results are estimated from a homogeneous variance model. If the test for homogeneity of variance is rejected (i.e., Test 2; $p < 0.05$), the model is run again while modeling the variance as a power function of the mean to account for this nonhomogeneous variance. If this nonhomogeneous variance model does not adequately fit the data (i.e., Test 3; $p < 0.05$), the data set is considered unsuitable for BMD modeling. Among all models providing adequate fit for a given endpoint, the benchmark dose lower confidence limit (BMDL) from the model with the lowest Akaike's information criterion (AIC) was selected as a potential POD when BMDL values were sufficiently close (within 3-fold). Otherwise, the lowest BMDL was selected as a potential POD for each endpoint.

Where modeling was feasible, the estimated BMDLs were used as PODs. Further details, including the modeling output and graphical results for the model selected for each endpoint, can be found in Supplemental Information, Appendix B. The benchmark dose approach involving modeling to obtain the BMDL is preferred, but it involves modeling dose levels corresponding to BMR levels near the low end of the observable range of the data and is not always feasible. When data sets were not amenable to BMD modeling, no-observed-adverse-effect level (NOAEL) or lowest-observed-adverse-effect level (LOAEL) values were selected and used as the POD based on expert judgment, considering the study design features (e.g., severity and rarity of the outcome; biological significance, considering the magnitude of change at the NOAEL or LOAEL; statistical significance and power; exposure and outcome ascertainment methods).

For the study by Iwai and Hoberman (2014), the experiment was conducted in two phases. Except for differences in the dose levels, the design and conduct were the same across the two phases. Specifically, in addition to concurrent control groups for each phase, animals were exposed

to 100, 350, or 500 mg/kg-day in Phase 1 and 7, 35 or 175 mg/kg-day in Phase 2. When possible, the two phases were combined for modeling to provide a more robust dose range. If the combined data set did not result in adequate model fit, the phases were modeled separately and the results for the individual phases were presented.

Approach for Animal-Human Extrapolation of PFHxA Dosimetry

The PFAS protocol (Appendix A) recommends the use of physiologically based pharmacokinetic (PBPK) models as the preferred approach for dosimetry extrapolation from animals to humans, while allowing for the consideration of data-informed extrapolations (such as the ratio of serum clearance values) for PFAS that lack a scientifically sound and sufficiently validated PBPK model. If chemical-specific information is not available, the protocol then recommends that doses be scaled allometrically using body weight $(BW)^{3/4}$ methods. This hierarchy of recommended approaches for cross-species dosimetry extrapolation is consistent with EPA's guidelines on using allometric scaling for deriving oral reference doses (U.S. EPA, 2011). This hierarchy preferentially prioritizes adjustments that result in reduced uncertainty in the dosimetric adjustments (i.e., preferring chemical-specific values to underpin adjustments versus use of default approaches).

As discussed in Section 3.1.5, no PBPK model is available for PFHxA in rats, mice, or monkeys. Although a PBPK model for humans was described by Fàbrega et al. (2015), it was not considered sufficiently reliable for use in an IRIS Toxicological Review.

There are, however, pharmacokinetic information available for PFHxA in relevant animal species (rats, mice, and monkeys) or humans that were useful for data-informed extrapolation approach for estimating the dosimetric adjustment factor (DAF) for PFHxA. Various PK analyses can be performed to extract meaningful information from PK data. Because PK data for various PFAS are available, including for PFBA (Chang et al., 2008), PFBS (Olsen et al., 2009), PFHxA (Dzierlenga et al., 2019), PFHxS (Sundström et al., 2012), PFNA (Tatum-Gibbs et al., 2011), and PFOA and PFOS (Kim et al., 2016b), that show a clear biphasic elimination pattern indicative of distinct distribution and elimination phases, EPA chose to use a two-compartment PK model, similar to the analysis of (Fujii et al., 2015). The EPA model is characterized by equation 5-1:

$$C(t) = A \cdot \exp(-\alpha \cdot t) + B \cdot \exp(-\beta \cdot t) - flag_{oral} \cdot (A+B) \exp(-k_a \cdot t), \quad (5\text{-}1)$$

where α and β are first-order rate constants (units of $time^{-1}$) representing the rate of distribution and elimination, respectively, k_a is a rate constant (units of $time^{-1}$) for oral absorption, and $flag_{oral}$ is set to zero when analyzing intravenous dose data or one for oral data. Details of the model fitting are provided in Appendix B. The model assumes that oral bioavailability is 100%, consistent with PK data from Dzierlenga et al. (2019) and other studies and that internal dosimetry and elimination are linear with dose. This is implicitly a two-compartment PK model represented by the model, for which the rate of elimination corresponds to β. It is presumed that the total

concentration from several consecutive doses would be obtained by simply adding the individual concentration curves, given the distinct dose times.

This PK model assumes the parameters are independent of time and dose. As discussed in the "Elimination" section, PK studies that measured tissue concentrations after multiple days of exposure are consistent with simple PK models parameterized from one-day exposure and support the assumption that the model parameters are independent of time. Although PK data at lower doses do not show any trend consistent with dose-dependence, data for the highest dose indicate that elimination can be reduced (Dzierlenga et al., 2019); the opposite of what is predicted based on the hypothesis of saturable resorption. While saturation of reabsorption transporters would lead to a decreased half-life at higher doses, there are also transporters responsible for elimination of PFAS to urine, such as Oat1 and Oat3, and saturation of these transporters could lead to an increase in observed half-life. A systematic deviation from the assumption of equal or more rapid clearance at higher doses has not been observed in the other relevant data (Iwabuchi et al., 2017; Gannon et al., 2011; Chengelis et al., 2009a). Further, because PFHxA is not metabolized, nonlinearity in its internal dose is not expected due to that mechanism. Parameter estimation, however, was performed both including and excluding the highest dose data. Had the resulting estimate of β been significantly different when the high-dose data were included, this would have indicated a dose dependence. The results of the alternative analyses did not indicate such a difference, however, leading to the conclusion that PFHxA PK is not dose dependent and that the assumption of nonvarying parameters in the PK model equation is appropriate. Further details are provided in Appendix C.

Given the fit of this model to a specific data set, the AUC from the time of exposure to infinity is:

$$AUC_{inf} = A/\alpha + B/\beta - flag_{oral} \cdot (A+B)/k_a \quad (5\text{-}2)$$

AUC is the integral of the chemical concentration in blood or serum over time, with units of mass × time / volume (e.g., mg-hr/L), and is considered an appropriate measure of internal dose when the chemical has an accumulative effect over time.

By definition, the clearance (CL) of a compound is the effective volume of blood cleared of the compound per unit time (units of volume/time). Mathematically, given the PK model described above, CL = dose/AUC_{inf}. If one assumes that risk increases in proportion to AUC, the ratio of clearance in animal to that in the human, CL_A:CL_H, can then be used to convert an oral dose-rate in animals (mg/kg-day) to a human equivalent dose (HED) rate. A similar approach using the ratio of the beta-phase half-lives can be used and is outlined in Appendix C, but that approach ignores differences in the absorption rate and alpha-phase distribution rate that impact AUC and is, therefore, considered to produce a more uncertain outcome. The relationship between the volume of distribution (Vd), CL and half-life ($t_{1/2}$) is given by:

$CL = \ln(2) \times Vd/T_{1/2}$.

Hence: $CL_A/CL_H = (Vd_A/Vd_H)/(t_{1/2,A}/t_{1/2,H})$.

So, if one assumes $Vd_H = Vd_A$, as the EPA has done, then:

$CL_A/CL_H = 1/(t_{1/2,A}/t_{1/2,H})$;

i.e., the clearance ratio is equal to one over the half-life ratio, so there is no quantitative difference between the two options for HED calculation in this case. However, if there were independent data to identify Vd_H, the value might not be identical to Vd_{rat}, in which case the clearance ratio would be different from the half-life ratio. Further, using the estimated clearance ratio makes the calculation more transparent because it requires an explicit statement of the assumption that $Vd_H = Vd_{rat}$, while use of the ratio of half-lives makes this assumption implicitly, hiding it from consideration as a factor introducing uncertainty.

Therefore, the HED was explicitly calculated using the ratio of clearance values:

$$HED = (CL_H/CL_{A[s]}) \times POD \quad (5\text{-}3)$$

Given the PK model and definition of clearance above, the resulting HED is the dose that results in the same AUC in humans as is predicted in animals exposed at the POD, if one can obtain a value of CL_H.

In the term $CL_{A[s]}$, the [s] in the subscript refers to the sex-specific value available for animals but not humans in the case of PFHxA. Because there are sex-specific values (significant differences between males and females) in clearance among mice and rats, the CL values for female rodents would be used to extrapolate health effects in female rodents and the CL values for male rodents would be used to extrapolate male rodent health effects. This choice simply ensures that an observed effect in male rats, for example, is extrapolated using the expected internal dose for male rats. When endpoints from both male and female animals are analyzed (i.e., separate dose-response analyses are conducted for results in males vs. females) resulting in sex-specific PODs, the corresponding male and female human HEDs would be calculated, using $(CL_H/CL_{A[s]})$.

The volume of distribution in the beta phase (i.e., after the chemical has distributed into the body as a whole) given the two-compartment model above is:

$$V_{d,\beta} = CL/\beta = dose/[\beta \times (A/\alpha + B/\beta - flag_{oral} \times (A+B)/k_a)] \quad (5\text{-}4)$$

Except for the i.v. dose data from Dzierlenga et al. (2019), the V_d for rats for all other experiments and studies for male and female rats were between 0.9 and 1.7 L/kg and the averages for males and females were virtually indistinguishable: 1.37 and 1.35 L/kg, respectively. For the i.v. dose data from Dzierlenga et al. (2019), $V_{d,\beta}$ was 5.2 L/kg in male rats and 18.7 L/kg in female rats.

In contrast, $V_{d,\beta}$ for the i.v. dose data from Chengelis et al. (2009a) was 0.93 L/kg for both male and female rats. Thus, excluding those specific i.v. experiments, $V_{d,\beta}$ in rats does not appear to be sex specific and an overall average of 1.36 L/kg appears appropriate for that species.

For male and female mice, the corresponding V_d was 0.75 and 0.78 L/kg, respectively, based on data from Gannon et al. (2011), again not indicating a significant sex difference, although the value is somewhat lower than in rats.

For male and female monkeys, Chengelis et al. (2009a) reported V_d = 0.99 ± 0.58 L/kg and 0.47 ± 0.35 L/kg, respectively. Although these indicate a possible sex difference, only three animals of each sex were used and the estimated ranges (0.39–1.5 vs. 0.23–0.87 L/kg) significantly overlap. Hence, some caution in interpreting these data is required. The overall average V_d for monkeys, 0.73 L/kg, is similar to the value for mice, although also lower than the value in rats.

Because the volume of distribution (V_d) has not been determined in humans, but an estimate for the human half-life ($t_{1/2}$) is available, three options for estimating a clearance in humans can be considered, although this might be viewed as extreme for the purpose of predicting HED values. The observed $t_{1/2}$ in humans is presumed to represent the beta or clearance phase, given the PFHxA study participant evaluation occurred over months after primary exposure to PFHxA had ended (Nilsson et al., 2010). Hence it is presumed that $t_{1/2} = \ln(2)/\beta$. Rearranging the two equations, $CL = V_{d,\beta} \times \beta = V_{d,\beta} \times \ln(2)/ t_{1/2}$. Three options were considered, as follows:

1) The V_d for humans is equal to that determined in the next closest species biologically, monkeys. This assumes the biological and biochemical factors that determine the tissue: serum concentration ratio and the relative proportion (fraction of BW) for various tissues is similar in humans and monkeys. This assumption presumes the relative binding of PFHxA in human serum relative to various other tissues in the body is like that in monkeys but leads to a conclusion that renal clearance in humans is significantly slower than in other species.

2) Use the clearance values estimated for mice, rats, and monkeys to estimate the clearance in humans via allometric scaling. The results for mice, rats, and monkeys in Table 5-3 show almost no trend with increasing species BW but can be fitted with a power function to obtain $CL = 0.152 \cdot BW^{-0.023}$ (L/kg), assuming standard BW values of 0.03 and 0.25 kg for mice and rats, respectively, and the reported BW of monkeys used by Chengelis et al. (2009a). For a standard human BW of 80 kg, the resulting predicted clearance in humans is 0.137 L/hr-kg. If this is the actual clearance in humans, but $t_{1/2}$ = 275 hr, human $V_{d,\beta} = CL \times t_{1/2}/\ln(2)$ = 54 L/kg. Note that human participants were exposed to PFHxA for months, which could have allowed them to accumulate a deep tissue dose, while the monkey PK study involved only a single i.v. administration. Thus, a much higher V_d might have been estimated in monkeys had they been subject to repeated doses.

3) The apparent human half-life estimated by EPA from the data of Nilsson et al. (2013) might be an artifact of significant ongoing exposure to PFHxA during the period of observation. Pérez et al. (2013) detected PFHxA levels in human tissues higher than other PFAS and other observational studies regularly detect PFHxA in human serum demonstrating widespread human exposure to the general population. Thus, there is no reason to believe the subjects of Nilsson et al. (2013) did not also have some level of ongoing exposure; the

question is whether such exposure was significant relative to the body burden accumulated from exposure as ski-wax technicians. If the value of CL estimated in (2) (0.137 L/hr-kg) is an accurate prediction for humans *and* the V_d is equal to the average estimated for monkeys (0.73 L/kg), the half-life in humans should be
$t_{1/2}$ = ln(2) × V_d /CL =ln(2) × 0.73 (L/kg)/(0.137 L/hr-kg) = 3.7 hours. If this were the case, human serum levels would fall 99% in a single day, while the data of Nilsson et al. (2013) show that such a decline takes at least 2 months and, even after a day or two off work, a technician's serum concentration would be near zero. Further, the serum concentrations reported Nilsson et al. (2013) do decline to near or below the limit of detection by late spring or early summer, indicating that other ongoing sources of exposure were not significant for that population. Thus, this third option seems extremely unlikely and was not evaluated further.

The two options for human CL estimated above are provided in Table 5-3.

Table 5-3. Summary of serum half-lives and estimated clearance for PFHxA

Species/sex	Study design	Elimination half-life ($t_{1/2}$) (hr)	Clearance (CL) (L/hr-kg)	Volume of distribution (V_d) (L/kg)	References/data sources
Rat, female	Oral and i.v.	2.7 (0.5–11.2)	0.383 (0.259–0.574)[a]	1.48 (0.27–4.42)[a]	Dzierlenga et al. (2019); Chengelis et al. (2009a); Gannon et al. (2011)
Rat, male	Oral and i.v.	5.4 (1.6–19.5)	0.163 (0.112–0.228)[a]	1.31 (0.37–4.4)[a]	Dzierlenga et al. (2019); Chengelis et al. (2009a); Iwabuchi et al. (2017); Gannon et al. (2011)
Mouse, female	Oral	7.9 (2.8–23)	0.206 (0.137–0.308)[a]	2.46 (0.82–6.82)[a]	Gannon et al. (2011); Daikin Industries (2010)
Mouse, male	Oral	10.6 (2.3–29)	0.0894 (0.053–0.153)[a]	1.38 (0.31–3.73)[a]	Gannon et al. (2011)
Monkey, female	i.v.	2.4	0.136	0.474 ± 0.349[b]	Chengelis et al. (2009a)
Monkey, male	i.v.	5.3	0.122	0.989 ± 0.579[b]	Chengelis et al. (2009a)
Human, male and female (data derived)	Ecological	275 (145–510)	1.84×10^{-3} [c] ($(0.45 – 7.35) \times 10^{-3}$)[d]	0.73(0.33–1.45)[e]	Nilsson et al. (2013)
Human, male and female (allometric)	NA	275	0.137[f]	54[f]	Nilsson et al. (2013)

[a]For each experiment (study/route/dose), a separate distribution of CL = dose/AUCinf and $V_d\beta$ = CL/β was generated. Median, 5th, and 95th percentiles of each distribution were calculated and are available on request. Results across experiments/dose levels were pooled, and the values presented here are statistics for the pooled results, 50th (5th–95th) percentiles for each species/sex.

[b]Reported mean ± SD from three male or female monkeys.

[c]CL = $V_d \times \ln(2)/t_{1/2}$ with V_d assumed as the average of the estimated values for male and female monkeys and $t_{1/2}$ estimated as described in Appendix C.2.

[d]As described in Section 3.1.4, the 90% CI for the elimination constant, $k_e = \ln(2)/t_{1/2}$, was combined with the estimated range of V_d to estimate an overall range of human CL.

[e]Upper and lower bounds of V_d for humans set to average value in rats (lower bound) and highest estimated individual value for monkeys (see Human Studies in Section 3.1.4), respectively.

[f]Human CL estimated by allometric scaling from values estimated for mice, rats, and monkeys; human $t_{1/2}$ from EPA estimate (Section 3.1.4); human $V_d = CL \times t_{1/2}/\ln(2)$.

Thus, two alternative values of the DAF, CL_H:$CL_{A[s]}$—which is the ratio of clearance values—can be obtained (see Table 5-4). Even though there is no apparent difference in CL between men and women, the sex difference in rats and mice means that the internal dose in male rats at a given applied dose, for example, will be different from the internal dose in female rats given the same dose. Use of DAFs specific to the sex of the experimental animals in which an endpoint is observed is needed to account for this difference in dosimetry among laboratory animals. The HED for 10 mg/kg-day in male rats is expected to be different from the HED for 10 mg/kg-day in female rats because the internal dose in male rats will be higher than the internal dose in female rats at that dose.

Table 5-4. Two options for rat, mouse, and human clearance values and data-informed dosimetric adjustment factor

Sex	Species	Animal clearance (L/hr-kg)[a]	Human clearance (L/hr-kg)	DAF (CL_H:$CL_{A[s]}$)
Male	Rat	0.163	1.84×10^{-3} [b] $((0.45 – 7.35) \times 10^{-3})$[c] (mean, estimated rage, using preferred [data-driven] approach)	1.1×10^{-2} $(2.8 \times 10^{-3} – 4.5 \times 10^{-2})$[d]
	Mouse	0.0894		2.1×10^{-2} $(5.0 \times 10^{-3} – 8.2 \times 10^{-2})$[d]
Female	Rat	0.383		4.8×10^{-3} $(1.2 \times 10^{-3} – 1.9 \times 10^{-2})$[d]
	Mouse	0.206		8.9×10^{-3} $(2.2 \times 10^{-3} – 3.6 \times 10^{-2})$[d]
Male	Rat	0.163	0.137[e] (alternative approach)	0.84
	Mouse	0.0894		1.5
Female	Rat	0.383		0.36
	Mouse	0.206		0.67

Shaded values were applied to derive the POD_{HED}.
[a]Species/sex-specific CL values (see Appendix C).
[b]Calculated from mean human elimination constant, k_e, obtained by Bayesian PK analysis and average volume of distribution for male and female monkeys (see Table 5-3).
[c]Calculated from 90% credible interval for the human elimination constant, k_e, obtained by Bayesian PK analysis and the estimated range of volume of distribution for humans (see Table 5-3).
[d]DAF uncertainty calculated from estimated range of human clearance value.
[e]Calculated from allometric scaling of CL using results in Table 5-3.

To evaluate whether it is more reasonable to expect CL or V_d to be similar in humans as in experimental animals, values of CL were examined directly in humans for PFHxS, PFNA, and PFOA by Zhang et al. (2013b) and can be compared to those for experimental animals. By comparing human and rat clearance for a set of compounds from the same chemical family, for which data are available in both species, a "read across" can be done to evaluate the most likely case for PFHxA. Note that PFHxS has the same carbon chain length as PFHxA (C_6) and while PFOA and PFNA have

longer chains (C_8 and C_9 respectively) they are still much more chemically similar to PFHxA than any other compounds for which corresponding human data are available. Briefly, Zhang et al. (2013b) measured PFAS concentrations in serum and matched 24-hour urine samples to directly measure urinary clearance. To avoid the complicating issue of losses from menstrual blood, results for men and women over the age of 50 years are evaluated. Median urinary CL values reported by Zhang et al. (2013b) were 0.015, 0.094, and 0.19 mL/kg-day for PFHxS, PFNA, and total PFOA (all isomers), respectively.

Kim et al. (2016b) reported renal PFHxS clearance of 0.76 mL/kg-day in rats while Kim et al. (2016b) and Sundström et al. (2012) reported *total* clearance of 7–9 mL/kg-day. Sundström et al. (2012) also reported total clearance of PFHxS of 3–5 mL/kg-day in male mice and 1.3–1.9 mL/kg-day in monkeys. Thus, these results for PFHxS show significantly slower clearance in humans than in mice, rats, and monkeys.

Dzierlenga et al. (2019) evaluated the PK of PFOA (as well as PFHxA) in male rats and obtained clearance values of 9–16 mL/kg/d, depending on the dose and route. Thus, PFOA is also cleared much more rapidly in rats than humans.

The reported dose/AUC can be used to derive clearance values for PFNA from the results of Tatum-Gibbs et al. (2011). The estimated CL in rats is highly variable across the studies evaluated but ranged from 2 to 66 mL/kg-day in males and from 4 to 106 mL/kg-day in females (Tatum-Gibbs et al., 2011; Benskin et al., 2009; De Silva et al., 2009; Ohmori et al., 2003). CL in male and female mice reported by Tatum-Gibbs et al. (2011) ranged from 3 to 10 mL/kg-day. Although the wide range for rats indicates a degree of uncertainty, these results indicate that clearance in mice and rats is similar and much larger than the corresponding human value (0.094 mL/kg-day) (Zhang et al., 2013b).

Thus, three other PFAS, including one with the same carbon-chain length as PFHxA, have been shown to have much lower clearance in humans than rats. Data for PFDA were not discussed here since it is a C_{10} compound, but it also shows a similar rat-human difference in clearance. Hence, a read-across analysis suggests that option (1) above is more likely to be true.

The alternative, option (2) above, requires one to accept that the V_d in humans is roughly two orders of magnitude higher than in rats and monkeys, although the biochemical factors that determine serum-tissue partitioning are expected to be conserved across mammalian species, as described in the section above on distribution. Hence, option (2) seems highly unlikely.

Therefore, the top set of DAFs in Table 5-4—based on CL_{human} = 1.84 × 10^{-3} L/kg-hr—are the preferred set because they are consistent with data for other PFAS, and the reasonable expectation, based on data from multiple chemicals, is the volume of distribution in humans does not substantially differ from that in experimental animals.

Representative calculations of the HED for considered health effects follow, using the POD of 20 mg/kg-day for postnatal (F_1) body weight at PND 0 (Loveless et al., 2009) as an example and the female rat DAF of 4.8 × 10^{-3}, based on clearance:

$$HED = POD\left(^{mg}/_{kg\text{-}day}\right) \times DAF$$
$$HED = 20\left(^{mg}/_{kg\text{-}day}\right) \times 4.8 \times 10^{-3} = 0.096\left(^{mg}/_{kg\text{-}day}\right) \quad (5\text{-}5)$$

In general, clearance captures the overall relationship between exposure and internal dose, specifically the average concentration of a substance in serum, while the half-life does not. Use of half-life makes an intrinsic assumption that V_d is the same in the test species as in humans. There is a significant difference between rats and monkeys, which leads to the expectation of a difference between rats and humans (see Table 5-3).

HED based on clearance incorporates the observed differences in V_d among mice, rats, and primates, and is therefore, the preferred approach for dosimetry extrapolation from animals to humans.

Uncertainty of Animal-human extrapolation of PFHxA dosimetry

Although the variability between, and even within, some data sets for rats (~4-fold for males and ~6-fold for females between the lowest and highest mean clearance values) is large, the number of studies provides confidence in the estimated average clearance values for both male and female rats, which is reflected by the modest 90% CI for rat CL in Table 5-3.

Only one PK study is available for mice, although with two dose levels (Gannon et al., 2011). Further, the data for the 100 mg/kg dose approach a plateau, as if clearance stopped when the concentration was around 0.5 µg/g, although such a plateau was not observed for the 2 mg/kg data. EPA concluded that the data, which used ^{14}C labeling, were not correctly adjusted for the background signal or LOD. EPA was able to analyze the two dose levels for male and female mice successfully, however, by focusing on the data above the concentration at which the plateau occurred. Because the data from Gannon et al. (2011) for rats is near the middle of the range for other rat studies and the methods described otherwise are appropriate, it is presumed that this study has good quality results, except for the LOD correction of this dose in mice, is presumed. Therefore, some uncertainty remains with the clearance value obtained for mice from this study.

The PK study of Chengelis et al. (2009a) is considered high quality, but the results for monkeys used only three males and three females.

Uncertainty in the application of the DAF based on clearance remains, given that neither V_d nor CL were measured or determined in humans. To estimate CL in humans, the human V_d was assumed equal to the average value estimated in male and female monkeys, which seems less uncertain given the data and analyses described above. The V_d of male and female mice was assumed the same as in male and female rats, respectively. Because the difference in V_d between male and female rats was small, using these sex-specific values for mice will give similar results to using an average.

One alternative approach to using clearance in mice or rats to estimate the average blood concentrations in those species for each bioassay might be to use the measured serum concentrations from toxicological studies as BMD modeling inputs and then the estimated human

clearance value to calculate the HED. Three of the four studies being evaluated, however, did not measure PFHxA serum concentrations (Klaunig et al., 2015; Iwai and Hoberman, 2014; Chengelis et al., 2009b; Loveless et al., 2009). Although Iwai and Hoberman (2014) attempted to measure serum concentrations in mice, all serum measurements were below the LOQ. Therefore, this alternative approach cannot be applied in evaluating these dose-response data.

There is uncertainty in the estimated human clearance because the V_d had to be extrapolated from animals (nonhuman primates) and the limited human PK data from only eight individuals with noncontrolled exposures. As discussed in Section 3.1.2, the distribution of PFHxA between serum and various tissues is determined by biochemical parameters such as the concentrations of various binding proteins and the affinity of PFHxA for those proteins, that are largely conserved across mammalian species. However, V_d values estimated for animals ranged between 0.33 L/kg in rats (Section 3.1.2) to 1.54 L/kg in one of six monkeys studied (Chengelis et al., 2009a), so uncertainty in the human V_d is presumed to be about 4.7-fold. Together with the estimated uncertainty in the human first-order elimination constant for which the 90% credible interval ranges 3.5-fold, an overall range of uncertainty in the human clearance of 16-fold (± 4-fold) was estimated (see Section 3.1.4 Pharmacokinetics-Elimination-Human Studies).

Application of Animal-Human Extrapolation for PFHxA Dosimetry

Table 5-5 presents the estimated POD_{HED} (mg/kg-day) values for the hepatic, developmental, and hematopoietic toxicity endpoints considered for RfD derivation based on the endpoint selection justification described above and in Table 5-1 and preferred DAF values presented in Table 5-4.

The last column in Table 5-5 includes normalization from the ammonium salt to the free acid using a molecular weight conversion [MW free acid/MW ammonium salt = 314/331 = 0.949 (Iwai and Hoberman, 2014)] and sodium salt to free acid [MW free acid/MW sodium salt = 314/336 = 0.935 (Loveless et al., 2009)]. The POD_{HED} for postnatal (F_1) body weights used the female HED, as exposures were to the dams and assumed equal clearance in a developing offspring as an adult.

The free acid of PFHxA is calculated using the ratio of molecular weights, as follows:

$$PFHxA\ (free\ acid) = \left(\frac{MW\ free\ acid}{MW\ ammonium\ salt}\right) = \left(\frac{314}{331}\right) = 0.949$$
$$PFHxA\ (free\ acid) = \left(\frac{MW\ free\ acid}{MW\ sodium\ salt}\right) = \left(\frac{314}{336}\right) = 0.935 \qquad (5\text{-}6)$$

Table 5-5. PODs considered for the derivation of the RfD

Endpoint	Study/confidence	Species, strain (sex)	POD type/model	POD (mg/kg-d)	POD_{HED} PFHxA[a] (mg/kg-d)
Hepatic effects					
↑Hepatocellular hypertrophy	Loveless et al. (2009) High confidence Subchronic (90 d)	Rat, Crl:CD(SD) (male)	BMDL10ER Multistage 1 NCV	10.66	0.11[c]
		Rat, Crl:CD(SD) (female)	NOAEL[b] (0% response)	100	0.45[c]
↑Hepatocellular necrosis	Klaunig et al. (2015) High confidence Chronic (104 w)	Rat, Crl:CD(SD) (female)	NOAEL[b] (5% response vs. 3.3% in controls)	30	0.14
Hematopoietic effects					
↓Hemoglobin	Klaunig et al. (2015) *High* confidence Chronic (51 w)	Rat, Crl:CD(SD) (female)	$BMDL_{1SD}$ Linear CV	122.77	0.59
	Chengelis et al. (2009b) *High* confidence Subchronic (90 d)	Rat, Crl:CD(SD) (male)	NOAEL[d] (7% decrease)	50	0.55
		Rat, Crl:CD(SD) (female)	NOAEL[d] (3% decrease)	50	0.24
	Loveless et al. (2009) *High* confidence Subchronic (90 d)	Rat, Crl:CD(SD) (male)	NOAEL[d] (6% decrease)	20	0.21[c]
		Rat, Crl:CD(SD) (female)	$BMDL_{1SD}$ Polynomial 3 CV	127.61	0.57[c]
↓Red blood cells	Klaunig et al. (2015) *High* confidence Chronic (51 w)	Rat, Crl:CD(SD) (male)	NOAEL[b] (4% decrease)	100	1.21
		Rat, Crl:CD(SD) (female)	$BMDL_{1SD}$ Linear CV	109.15	0.52
	Chengelis et al. (2009b) *High* confidence Subchronic (90 d)	Rat, Crl:CD(SD) (male)	NOAEL[d] (no change)	50	0.55
		Rat, Crl:CD(SD) (female)	$BMDL_{1SD}$ Exponential 5 CV	16.32	0.078
	Loveless et al. (2009) *High* confidence Subchronic (90 d)	Rat, Crl:CD(SD) (male)	$BMDL_{1SD}$ Linear NCV	44.57	0.46[c]
		Rat, Crl:CD(SD) (female)	$BMDL_{1SD}$ Linear CV	112.36	0.50[c]
Developmental effects					
↓Postnatal (F_1) body weight, PND 0	Loveless et al. (2009) *High* confidence	Rat, Crl:CD(SD), F_1 (combined)	$BMDL_{5RD}$ Hill	10.62	0.048[c]

Endpoint	Study/confidence	Species, strain (sex)	POD type/model	POD (mg/kg-d)	POD_{HED} PFHxA[a] (mg/kg-d)
↓Postnatal (F_1) body weight, PND 0	Iwai and Hoberman (2014) *High* confidence	Mouse, CD-1, F_1 (combined)	$BMDL_{5RD}$ Polynomial 3 CV Phase 2	80.06	0.68[e]
↓Postnatal (F_1) body weight, PND 4			$BMDL_{5RD}$ Exponential-M5 Phase 1 and 2 Polynomial 3 CV Phase 2	102.94 89.79	0.87[e] 0.76[e]
↑Perinatal (F_1) mortality (PND 0–21, including stillbirths)	Iwai and Hoberman (2014) *High* confidence	Mouse, CD-1, F_1 (combined)	$BMDL_{1E1R0}$ NLogistic Phase 2	24.77	0.21[e]
Endocrine effects					
↓Total T4	NTP (2018) *High* confidence Short term (28 d)	Rat, Harlan Sprague-Dawley (male)	$BMDL_{1SD}$ Hill CV	25.97	0.29

CV = constant variance; NCV = nonconstant variance; SD = standard deviation.

[a]HED calculations based on the DAF, the ratio of human and animal clearance values (see Table 5-4). DAF values for female rats and female mice were used for the respective developmental effects on combined male and female pups of each species. POD_{HED} based on PFHxA free acid.

[b]Response only at high dose with responses far above BMR level, data not modeled.

[c]POD_{HED} multiplied by normalization factor to convert from sodium salt to free acid (MW free acid/MW sodium salt = 314/336 = 0.935).

[d]No models provided adequate fit; therefore, a NOAEL approach was selected.

[e]POD_{HED} multiplied by normalization factor to convert from ammonium salt to the free acid (MW free acid/MW ammonium salt = 314/331 = 0.949).

Derivation of Candidate Toxicity Values for the RfD

As discussed below, the subchronic, chronic, developmental, and short-term studies for hepatic, hematopoietic, developmental, and endocrine effects after PFHxA exposure were considered for use in deriving candidate toxicity values for the RfD. The PODs presented in Table 5-5 were considered and specific PODs were advanced for candidate toxicity value derivation over others within each health outcome category based on the confidence in the study, endpoint sensitivity and specificity, the POD_{HED}, and other considerations including uncertainty (see Table 5-6), as discussed below. The candidate lifetime toxicity values are presented in Table 5-7.

For hepatic outcomes, candidate toxicity values for both increased hepatocellular hypertrophy and necrosis were derived. From the PODs for hepatocellular hypertrophy (Loveless et al., 2009), the POD for male rats was selected was selected over the POD for female rats as the POD_{HED} from males was several-fold lower. Although. Although increased hepatocellular hypertrophy was considered as likely to be precursor event of more severe outcomes such as

necrosis (which might warrant deprioritizing necrosis), a candidate toxicity value for necrosis was also derived as both studies were of *high* confidence and the POD_{HEDs} did not notably differ.

For hematopoietic outcomes, data on RBCs and HGB from three *high* confidence studies were advanced for estimation of POD_{HEDs}. Of those, the lowest POD_{HED}, by a large margin, was for decreased RBCs in female rats from the *high* confidence subchronic study (Chengelis et al., 2009b) (POD_{HED} = 0.078 mg/kg-d). A candidate value for this POD was derived. Given the sensitivity of this POD over other PODs from the two subchronic studies and no clear reason to advance the other PODs (e.g., both studies were of *high* confidence and RBCs and HGB were interpreted to have similar sensitivity and specificity), none of the other PODs for RBCs or HGB from the subchronic studies were used to calculate candidate values. However, some caution in this POD is carried forward to candidate value derivation as the unadjusted POD was far below the observed NOAEL (50 mg/kg-d), suggesting that variability in the data may be a drive the candidate value lower. The subchronic POD_{HED} of 0.078 mg/kg-d (Chengelis et al., 2009b) was also 7-fold lower than the RBC finding available from the chronic study observed in the same sex, species, dose, and the magnitude of change in the response at 200 mg/kg-d was also similar (~8% decrease RBC). Although both subchronic and chronic exposure designs and study durations include the life cycle of a red blood cell (~60 days in rats), the subchronic study duration design may miss longer term effects on RBC regeneration. Therefore, although the POD_{HED} was less sensitive, the chronic POD for RBCs was also advanced for candidate toxicity value derivation.

For developmental outcomes, the *high* confidence finding of decreased postnatal (F_1) body weight in rats at PND 0 had the lowest POD_{HED} of 0.048 mg/kg-d (Loveless et al., 2009). This was 4-fold lower than the next lowest POD_{HED} for increased perinatal mortality in mice (Iwai and Hoberman, 2014), and considerably lower than POD_{HEDs} for offspring body weight decreases in mice. Ultimately, given the large difference in PODs for body weight changes, the PODs in mice were not advanced and only a candidate toxicity value for decreased postnatal (F_1) body weight in rats was derived. Because derived. Because changes in fetal body weight are less severe than fetal mortality and the POD_{HED} for the fetal body weight change in rats was only 4-fold more sensitive to PFHxA exposure than the POD_{HED} for offspring mortality in mice and therefore a candidate toxicity value was also derived for the increases in perinatal mortality in mice (Iwai and Hoberman, 2014).

For endocrine effects, the available POD was limited to thyroid hormone data from a short-term study in rats (NTP, 2018). Consistent with EPA's *A Review of the Reference Dose and Reference Concentration Processes* (U.S. EPA, 2002c) and for the purposes of this assessment, a lifetime toxicity value is not supported given the high degree of uncertainty when using PODs from a short-term study to protect against effects of exposure for a lifetime. The general lack of long-term or developmental studies of endocrine effects introduces uncertainty, as discussed below.

Under EPA's A Review of the Reference Dose and Reference Concentration Processes (U.S. EPA, 2002c) and Methods for Derivation of Inhalation Reference Concentrations and Application of Inhalation Dosimetry (U.S. EPA, 1994), five possible areas of uncertainty and variability were

considered in deriving the candidate toxicity values for PFHxA. An explanation of these five possible areas of uncertainty and variability and the UF values assigned to each of the of the POD_{HED} values selected for use in deriving candidate toxicity values, as well as the rationales for these decisions are listed in Table 5-6.

Table 5-6. Uncertainty factors for the development of the RfD for PFHxA

UF	Value	Justification
UF_A	3	As described in EPA's *A Review of the Reference Dose and Reference Concentration Processes* (U.S. EPA, 2002c), the interspecies uncertainty factor (UF_A) is applied to account for extrapolation of animal data to humans. A UF_A of 3 is applied to account for uncertainty in characterizing the PK and pharmacodynamic differences between species (i.e., from rats or mice to humans) following oral PFHxA exposure. Some aspects of the cross-species extrapolation of PK processes have been accounted for by calculating an HED through application of a DAF based on animal and human clearance; however, residual uncertainty related to potential pharmacodynamic differences remains. Typically, a threefold UF is applied for this uncertainty in the absence of chemical-specific information. This is the case for the hepatic, hematopoietic, and developmental effects of PFHxA.
UF_H	10	A UF_H of 10 is applied for interindividual variability in humans in the absence of quantitative information on potential differences in PK and pharmacodynamics relating to PFHxA exposure in humans.
UF_S	1 (developmental; hepatic and hematopoietic [chronic study])	A UF_S of 1 is applied to the developmental endpoint from the one-generation reproductive study by Loveless et al. (2009) and developmental study by Iwai and Hoberman (2014). The developmental period is recognized as a susceptible lifestage and exposure designs capturing sensitive developmental windows (e.g., gestation) are more relevant for the induction of developmental effects than lifetime exposures (U.S. EPA, 1991). Although effects on body weights are not unique to development, the current evidence for PFHxA suggests this is a sensitive lifestage for body weight effects of PFHxA exposure based on effects being measured at lower doses than adults. A UF_S of 1 is also applied to hematopoietic (i.e., decreased RBC) and hepatic (i.e., necrosis) endpoints in the chronic study (Klaunig et al., 2015) as the 51 wks of daily exposure is considered sufficiently representative of exposure for the rodents' lifetime and the incidence or severity of these outcomes is not anticipated to increase with longer exposure duration.

UF	Value	Justification
	3 (hepatic and hematopoietic [subchronic study])	A UF_S of 3 is applied to hepatocellular hypertrophy for the purpose of deriving a lifetime RfD. Although the endpoint was derived from a 90-d subchronic study (Loveless et al., 2009), which would typically warrant application of a UF_S = 10, there are some other sparse data that mitigate this uncertainty, to an extent. Specifically, significant hepatocellular hypertrophy was not observed in the chronic study in male or female rats (Klaunig et al., 2015). However, a UF_S = 1 was not applied as the evidence supports a pathway where hepatocellular hypertrophy is an adverse event leading to more severe outcomes with longer exposure durations, such as the necrosis that was observed in female rats in the chronic study. Additionally, the highest dose levels used in the chronic study were at or below the LOAEL for this effect in the available subchronic studies (see Section 3.2.1). Thus, some uncertainty remains and a UF_S of 3 is applied. A UF_S of 3 is applied to the hematopoietic endpoint (i.e., decreased RBCs) from the 90-d subchronic study (Chengelis et al., 2009b). Specifically, a UF_S lower than 10 was warranted as more significant effects on RBCs were not observed after chronic exposure at the same PFHxA doses (RBCs decreases of the same magnitude were observed at matched doses and sexes across exposure durations see Section 3.2.4); however, uncertainty remains when considering the doses tested in the chronic as compared to the subchronic study. Further, the subchronic study may poorly predict a chronic exposure setting across multiple RBC life cycles (one cycle is ~60 d), which could reflect cumulative effects as greater proportions of RBCs across stages are affected, or possibly even reduced effects (compensatory responses) warranting a UF_S higher than 1. Thus, a UF_S of 3 was applied.
UF_L	1	A UF_L of 1 is applied for LOAEL-to-NOAEL extrapolation when the POD is a BMDL or a NOAEL.
UF_D	3	The database uncertainty factor (UF_D) is applied to account for the potential of deriving an under-protective reference value as a result of incomplete characterization of a chemical's toxicity (U.S. EPA, 2002c). For PFHxA, a UFD of 3 was selected to account for deficiencies and uncertainties in the database. Although not large, the available evidence base spans a number medium and high confidence studies in laboratory animals, including several short-term studies, two subchronic studies, and one chronic study in Sprague-Dawley rats, as well as developmental/reproductive studies in Sprague-Dawley rats and Crl:CD1 mice. Limitations in the database, as described in U.S. EPA (2002c) were used as the basis for a UF_D = 3. These limitations included a lack of informative human studies for most outcomes; subchronic or chronic toxicity studies in more than one species; studies of potential multigenerational effects; effects; developmental neurotoxicity studies; and thyroid toxicity studies after PFHxA exposure during development or after long-term exposure.
UF_C	See Table 5-7 and Table 5-11	Composite uncertainty factor = $UF_A \times UF_H \times UF_S \times UF_L \times UF_D$.

UF_A = interspecies uncertainty factor, UF_H = interindividual variability in humans uncertainty factor, UF_S = extrapolating from subchronic to chronic uncertainty factor, UF_L = LOAEL-to-NOAEL extrapolation uncertainty factor, UF_D = database uncertainty factor.

The uncertainty factors described in Table 5-6 were applied and the resulting candidate values for use in estimating an RfD for lifetime exposure are shown in Table 5-7.

Table 5-7. Candidate values for PFHxA

Endpoint/study/ confidence	Species, strain (sex)	POD_{HED} PFHxA[a] (mg/kg-d)	UF_A	UF_H	UF_S	UF_L	UF_D	UF_C	Candidate value PFHxA (mg/kg-d)	Candidate value PFHxA-Na[b] (mg/kg-d)	Candidate value PFHxA-NH_4[b] (mg/kg-d)
Hepatic effects											
↑Hepatocellular hypertrophy, 90 d Loveless et al. (2009) *High* confidence	Rat, Crl:CD(SD) (male)	0.11	3	10	3	1	3	300	4×10^{4}	4×10^{-4}	4×10^{-4}
↑Hepatocellular necrosis 2 y Klaunig et al. (2015) *High* confidence	Rat, Crl:CD(SD) (female)	0.144	3	10	1	1	3	300	5×10^{-4}	5×10^{-4}	5×10^{-4}
Hematopoietic effects											
↓Red blood cells, 51 wks Klaunig et al. (2015) *High* confidence	Rat, Crl:CD(SD) (female)	0.52	3	10	1	1	3	100	5×10^{-3}	6×10^{-3}	5×10^{-3}
↓Red blood cells 90 d Chengelis et al. (2009b) *High* confidence	Rat, Crl:CD(SD) (male)	0.078	3	10	3	1	3	300	3×10^{-4}	3×10^{-4}	3×10^{-4}
Developmental effects											
↓F_1 body weight, PND 0 Loveless et al. (2009) *High* confidence	Rat, Sprague-Dawley, F_1 (combined)	0.048	3	10	1	1	3	100	5×10^{-4}	5×10^{-4}	5×10^{-4}
↑Perinatal (F_1) mortality (PND 0–21, including stillbirths) Iwai and Hoberman (2014) High confidence	Mouse, CD-1, F_1 (combined)	0.21	3	10	1	1	3	100	2×10^{-3}	2×10^{-3}	2×10^{-3}

[a]HED calculations based on DAF, the ratio of human and animal clearance values (see Table 5-4). DAF values for female rats and female mice were used for the respective developmental effects on combined male and female pups of each species.

[b]To calculate candidate values for salts of PFHxA, multiply the candidate value of interest by the ratio of molecular weights of the free acid and the salt. For example, for the sodium salt of PFHxA, the candidate value would be calculated by multiplying the free acid candidate value by 1.070 (MW free acid/MW sodium salt = 336/314 = 1.070). This same conversion can be applied to other salts of PFHxA, such as the ammonium salt.

Selection of Lifetime Toxicity Value(s)

Selection of Organ- or System-Specific RfDs

Organ/system-specific (os)RfDs associated with each health effect are presented in Table 5-8 as they could be useful for certain decision purposes (i.e., site-specific risk assessments). The rationale for and application of osRfD are described in the PFAS Protocol, Appendix A. Confidence in each osRfD is described in Table 5-8 and is based on several factors, including confidence in the study, the evidence base supporting the hazard, and quantitative estimate for each osRfD.

The candidate toxicity value of 4×10^{-4} mg/kg-d PFHxA for hepatocellular hypertrophy was selected as the hepatic osRfD. Considering that hepatocellular hypertrophy likely precedes necrosis and is a slightly more sensitive endpoint than necrosis, hepatocellular hypertrophy from male rats in the subchronic study (Loveless et al., 2009) was selected over the candidate value for necrosis.

The candidate toxicity value of 5×10^{-3} mg/kg-d PFHxA for decreased RBCs in female rats from the chronic study was selected as the hematopoietic osRfD. Although the candidate value for the subchronic study (3×10^{-4} mg/kg-d PFHxA) was lower than that for the chronic study, confidence in the POD from the subchronic study was reduced as the unadjusted POD was well below the observed NOAEL in the study and may have been driven largely by variability in the data. Note that the unadjusted PODs for decreased RBCs in the other subchronic study datasets were more similar to the POD from the chronic study as shown in Table 5-7. Further, the subchronic study may poorly predict a chronic exposure setting across multiple RBC life cycles (one cycle is ~60 days), which could reflect cumulative effects as greater proportions of RBCs across stages are affected, or possibly even reduced effects (compensatory responses). Therefore, the value from the chronic study was interpreted as the most appropriate value for use in addressing the potential hematopoietic effects of lifetime exposure.

The candidate toxicity value of 5×10^{-4} mg/kg-d PFHxA for decreased F_1 offspring body weight at PND 0 rats (Loveless et al., 2009) was selected as the developmental osRfD. This candidate value was selected over the candidate value for increased perinatal mortality in mice (Iwai and Hoberman, 2014) because changes in fetal body weight are less severe and candidate value was substantially lower than fetal mortality. Therefore, this value is interpreted as protective of all developmental effects, including the increases in perinatal mortality in mice. In addition, of the two study designs, Loveless et al. (2009) included a longer exposure that spanned the entirety of gestation (exposure continued through the end of lactation, but the effect in question was measured on PND 0) versus Iwai and Hoberman (2014) where mouse offspring were exposed from GD 6–18; thus, the study in rats may encompass a greater proportion of the relevant critical windows for the developmental effects of PFHxA exposure.

Table 5-8. Confidence in the organ/system specific RfDs for PFHxA

Confidence categories	Designation	Discussion
Hepatic osRfD = 4×10^{-4} mg/kg-d PFHxA; 4×10^{-4} mg/kg-d PFHxA-Na and PFHxA-NH_4		
Confidence in the study used to derive osRfD	*High*	Confidence in the study (Loveless et al., 2009) is *high* based on the study evaluation results (i.e., rated high confidence overall) (HAWC link). The overall study size, design, and test species were considered relevant for deriving toxicity values.
Confidence in the evidence base for hepatic effects	*Medium*	Confidence in the oral toxicity evidence base for hepatic effects is *medium* based on consistent, dose-dependent, and biologically coherent effects on organ weight and histopathology observed in multiple *high* confidence subchronic and chronic studies. The available mechanistic evidence also supports biological plausibility of the observed effects. Limitations of the evidence base for hepatic effects is the lack of human studies, and measures that would have been useful to inform the pathways for hepatic effects leading to hepatocellular hypertrophy (e.g., specific histological stains for hepatic vacuole contents).
Confidence in quantification of the POD_{HED}	*Medium*	Confidence in the quantification of the POD and osRfD is *medium*, given the POD was based on BMD modeling within the range of the observed data and dosimetric adjustment based on PFHxA-specific PK information. Some residual uncertainty in the application of the dosimetric approach described above is that V_d and CL were not measured in humans or mice and were considered equivalent to those for monkeys and rats, respectively.
Overall confidence in the hepatic osRfD	*Medium*	The overall confidence in the osRfD is *medium* and is primarily driven by *medium* confidence in the overall evidence base for hepatic effects, *high* confidence in the study, and *medium* confidence in quantitation of the POD. *High* confidence in the study was not interpreted to warrant changing the overall confidence from *medium*.
Hematopoietic osRfD = 5×10^{-3} mg/kg-d PFHxA and PFHxA-NH_4; 6×10^{-3} mg/kg-d PFHxA-Na		
Confidence in study	*High*	Confidence in the study (Klaunig et al., 2015) is *high* based on the study evaluation results (i.e., rated *high* confidence overall) (HAWC link) and characteristics that make it suitable for deriving toxicity values, including relevance of the exposure paradigm (route, duration, and exposure levels), use of a relevant species, and the study size and design.
Confidence in evidence base for hematopoietic effects	*Medium*	Confidence in the evidence base for hematopoietic effects was *medium* based on consistent and biologically coherent effects on red blood cells, hemoglobin, and other hematological parameters measured across multiple *high* confidence chronic and subchronic studies. The RBC and hemoglobin findings were correlative with an erythrogenic response indicated by increased reticulocytes and pathological findings of splenic extramedullary hematopoiesis and bone marrow erythroid hyperplasia. Limitations of the hematopoietic evidence base are lack of human studies, and some hematological measures were observed only at the highest dose, limiting interpretation of dose-response.

Confidence categories	Designation	Discussion
Confidence in the quantification of the POD_{HED}	*Medium*	Confidence in the quantification of the POD and osRfD is *medium* given the POD was based on BMD modeling within the range of the observed data and dosimetric adjustment based on PFHxA-specific PK information. Some residual uncertainty in the application of the dosimetric approach described above is that V_d and CL were not measured in humans or mice and were considered equivalent to those for monkeys and rats, respectively. Some additional residual uncertainty in POD quantitation was due to the availability of a much lower POD, and the dose spacing was biased toward low end of the dose range.
Confidence in hematopoietic osRfD	*Medium*	The overall confidence in the osRfD is *medium* and is primarily driven by *medium* confidence in the overall evidence base for hematopoietic effects, and *medium* confidence in quantitation of the POD. *High* confidence in the study was not interpreted to warrant changing the overall confidence from *medium*.
Developmental osRfD = 5×10^{-4} mg/kg-d PFHxA; 5×10^{-4} mg/kg-d PFHxA-Na and $PFHxA-NH_4$		
Confidence in study	*High*	Confidence in the study (Loveless et al., 2009) is *high* based on study evaluation results (i.e., rated *high* confidence overall) (HAWC link) and characteristics that make it suitable for deriving toxicity values, including relevance of the exposure paradigm (route, duration, and exposure levels), use of a relevant species, and the study size and design.
Confidence in evidence base for developmental effects	*Medium*	Confidence in evidence base for developmental effects is *medium* based on the availability of data from two studies in different species (i.e., rats and mice) that consistently observed decreases in offspring body weight and coherent increases in perinatal mortality. Areas of uncertainty included lack of human data and multigenerational animal toxicity studies. Also, data to inform other organ/system-specific hazards (e.g., thyroid, immune, nervous system) following a developmental exposure are lacking. Together these present significant data gaps in the potential effects during this sensitive life stage.
Confidence in the quantification of the POD_{HED}	*Medium*	Confidence in the quantification of the POD and osRfD is *medium* given the POD was based on BMD modeling within the range of the observed data and dosimetric adjustment based on PFHxA-specific PK information. Some residual uncertainty in the application of the dosimetric approach described above is that V_d and CL were not measured in humans or mice and were considered equivalent to those in monkeys and rats, respectively.
Confidence in developmental osRfD	*Medium*	The overall confidence in the osRfD is *medium* and is primarily driven by *medium* confidence in the overall evidence base for developmental effects, *high* confidence in the study, and *medium* confidence in quantitation of the POD. *High* confidence in the study was not interpreted to warrant changing the overall confidence from *medium*.

Selection of the overall oral reference dose and confidence statements

Table 5-9. Organ/system specific RfD values for PFHxA

System	Basis	POD_{HED}	UF_C	osRfD for PFHxA (mg/kg-d)	osRfD for PFHxA-Na[a] (mg/kg-d)	osRfD for PFHxA-NH_4[a] (mg/kg-d)	Confidence
Hepatic	Increased hepatocellular hypertrophy in adult male Crl:CD Sprague-Dawley rats	0.11 mg/kg-d based on $BMDL_{10ER}$ and free salt normalization (Loveless et al., 2009)	300	4×10^{-4}	4×10^{-4}	4×10^{-4}	*Medium*
Hematopoietic	Decreased red blood cells in adult female Crl:CD Sprague-Dawley rats	0.52 mg/kg-d based on $BMDL_{1SD}$ (Klaunig et al., 2015)	100	5×10^{-3}	6×10^{-3}	5×10^{-3}	*Medium*
Developmental (selected as RfD)	Decreased postnatal (PND 0) body weight in F_1 Sprague-Dawley male and female rats, exposed throughout gestation and lactation	0.048 mg/kg-d based on $BMDL_{5RD}$ and free salt normalization (Loveless et al., 2009)	100	5×10^{-4}	5×10^{-4}	5×10^{-4}	*Medium*

[a]To calculate candidate values for salts of PFHxA, multiply the candidate value of interest by the ratio of molecular weights of the free acid and the salt. For example, for the sodium salt of PFHxA, the candidate value would be calculated by multiplying the free acid candidate value by 1.070 (MW free acid/MW sodium salt = 336/314 = 1.070). This same conversion can be applied to other salts of PFHxA, such as the ammonium salt.

From the identified human health effects of PFHxA and derived osRfDs for hepatic, hematopoietic, and developmental effects (see Table 5-9), an ***RfD of 5 × 10^{-4} mg/kg-day PFHxA based on decreased postnatal (F_1) body weight*** in rats was selected. As described in Table 5-8, confidence in the RfD is **medium**, based on medium confidence in the developmental RfD. The decision to select the developmental RfD was based on all available osRfDs in addition to overall confidence and composite uncertainty for those osRfDs. The confidence in the selected RfD is equivalent to that of the hepatic and hematopoietic RfDs. The developmental endpoint decreased F_1 body weight at PND 0 having the lowest overall POD_{HED} of 0.048 mg/kg-d PFHxA based on $BMDL_{5RD}$ and free salt normalization (Loveless et al., 2009) and UF_C of 100 was

considered protective across all lifestages. The hepatic RfD was slightly lower but was based on a higher POD_{HED} (0.11 mg/kg-day PFHxA) and UF_C (300). The developmental RfD, therefore, is based on the lowest POD_{HED} and lowest UF_C using a study considered *high* confidence.

Estimation or Selection of Points of Departure (PODs) for Subchronic RfD Derivation

In addition to providing an RfD for lifetime exposure in health systems, this document also provides an RfD for less-than-lifetime ("subchronic") exposures. These subchronic RfDs were based on the endpoints advanced for POD derivation provided in Table 5-1. Data to inform potential hepatic and hematopoietic effects from the *high* confidence subchronic studies by (Chengelis et al., 2009b; Loveless et al., 2009) were considered the most informative for developing candidate values. The *high* confidence developmental/reproductive studies (Iwai and Hoberman, 2014; Loveless et al., 2009) were also advanced for candidate value derivation. While it was not advanced for a lifetime RfD, the *high* confidence short-term study (NTP, 2018) was considered for subchronic candidate value derivation for endocrine effects. In general, the rationales for advancing these endpoints for subchronic value derivation are the same as described and summarized above in Table 5-1; however, for hematopoietic effects, subchronic data from Chengelis et al. (2009b) and Loveless et al. (2009) were prioritized over the data from the chronic study by Klaunig et al. (2015) for use in deriving a subchronic RfD.

The endpoints selected for dose-response were modeled using approaches consistent with EPA's *Benchmark Dose Technical Guidance* document (U.S. EPA, 2012a). The approach was the same as described above for derivation of lifetime toxicity values, the BMRs selected for dose-response modeling and the rationales for their selection (see Table 5-2), and the dosimetric adjustments using the ratio of the clearance in animal to that in the human and salt to free acid normalization. Table 5-10 presents the estimated POD_{HED} (mg/kg-day) values for the hepatic, developmental, and hematopoietic toxicity endpoints considered for subchronic RfD derivation.

Table 5-10. PODs considered for the derivation of the subchronic RfD

Endpoint	Study/confidence	Species, strain (sex)	POD type/model	POD (mg/kg-d)	POD_{HED} PFHxA[a] (mg/kg-d)
Hepatic effects					
↑Hepatocellular hypertrophy					
	Loveless et al. (2009) *High* confidence	Rat, Crl:CD(SD) (male)	$BMDL_{10ER}$ Multistage 1 NCV	10.66	0.11[c]
		Rat, Crl:CD(SD) (female)	NOAEL[b] (0% response)	100	0.45[c]

Endpoint	Study/confidence	Species, strain (sex)	POD type/model	POD (mg/kg-d)	POD_{HED} PFHxA[a] (mg/kg-d)
Hematopoietic effects					
↓Hemoglobin	Chengelis et al. (2009b) *High* confidence	Rat, Crl:CD(SD) (male)	$BMDL_{1SD}$ Polynomial 3 CV	81.35	0.89
		Rat, Crl:CD(SD) (female)	NOAEL[d] (3% decrease)	50	0.24
	Loveless et al. (2009) *High* confidence	Rat, Crl:CD(SD) (male)	NOAEL[d] (6% decrease)	202	0.21[c]
		Rat, Crl:CD(SD) (female)	$BMDL_{1SD}$ Polynomial 3 CV	127.61	0.57[c]
↓Red blood cell	Chengelis et al. (2009b) *High* confidence	Rat, Crl:CD(SD) (male)	NOAEL[d] (no change)	50	0.55
		Rat, Crl:CD(SD) (female)	$BMDL_{1SD}$ Exponential 5 CV	16.32	0.078
	Loveless et al. (2009) *High* confidence	Rat, Crl:CD(SD) (male)	$BMDL_{1SD}$ Linear NCV	44.57	0.46[c]
		Rat, Crl:CD(SD) (female)	$BMDL_{1SD}$ Linear CV	112.36	0.50[c]
Developmental Effects					
↓Postnatal (F_1) body weight, PND 0	Loveless et al. (2009) *High* confidence	Rat, Crl:CD(SD), F_1 (combined)	$BMDL_{5RD}$ Hill	10.62	0.048[c]
↓Postnatal (F_1) body weight, PND 0	Iwai and Hoberman (2014) *High* confidence	Mouse, CD-1, F_1 (combined)	$BMDL_{5RD}$ Polynomial 3 CV Phase 2	80.06	0.68[e]
↓Postnatal (F_1) body weight, PND 4			$BMDL_{5RD}$ Exponential-M5 Phase 1 and 2 Polynomial 3 CV Phase 2	102.94 89.79	0.87[e] 0.76[e]
↑Perinatal Mortality	Iwai and Hoberman (2014) *High* confidence	Mouse, CD-1, F_1 (combined)	$BMDL_{1ER}$ Nested Logistic Phase 2	24.77	0.21[e]

Endpoint	Study/confidence	Species, strain (sex)	POD type/model	POD (mg/kg-d)	POD_{HED} PFHxA[a] (mg/kg-d)
Endocrine effects					
↓ Total T4	NTP (2018) *High* confidence	Rat, Harlan Sprague-Dawley (male)	$BMDL_{1SD}$ Hill CV	25.97	0.29

1SD = 1 standard deviation, CV = constant variance, NCV = nonconstant variance.
[a]HED calculations based on the DAF, the ratio of human and animal clearance values (see Table 5-3). DAF values for female rats and female mice were used for the respective developmental effects on combined male and female pups of each species. POD_{HED} based on PFHxA free acid.
[b]Response only at high dose with responses far above BMR level, data not modeled.
[c]POD_{HED} multiplied by normalization factor to convert from sodium salt to free acid (MW free acid/MW sodium salt = 314/336 = 0.935).
[d]No models provided adequate fit; therefore, a NOAEL approach was selected.
[e]POD_{HED} multiplied by normalization factor to convert from sodium salt to free acid (MW free acid/MW ammonium salt = 314/331 = 0.949).

Derivation of Candidate Toxicity Values for the Subchronic RfD

The POD_{HED} values listed in Table 5-10 were further narrowed for selecting candidate toxicity values for subchronic osRfD derivation and subchronic RfD selection. As described for the RfD, RBCs were a more sensitive POD_{HED} for hematopoietic effects. Therefore, the red blood cell endpoint from female rats from Chengelis et al. (2009b) was advanced as the candidate toxicity value for subchronic RfD derivation over male endpoints for hematocrit and red blood cells based on RBC being more sensitive and therefore expected to be protective of effects in both sexes. Applying the rationales described for the selection of the lifetime osRfDs, the same endpoints were advanced as the candidate toxicity values for derivation of the hepatic and developmental subchronic osRfDs: male hepatocellular hypertrophy and decreased F_1 body weight at PND 0 (Loveless et al., 2009). For endocrine effects, a candidate toxicity value for the subchronic RfD was derived based on a short-term study showing decreased total T4 in adult male rats exposed for 28 days. Due to the high uncertainty associated with deriving a lifetime value based on a short-term study this endpoint was not considered for the RfD but was advanced for the subchronic RfD.

As described above under "Derivation of Candidate Values for the RfD," and in U.S. EPA (2002c), five possible areas of uncertainty and variability were considered in deriving the candidate subchronic toxicity values for PFHxA. In general, the explanations for these five possible areas of uncertainty and variability and the values assigned to each as a designated UF to be applied to the candidate POD_{HED} values are listed above and in Table 5-6, including the UF_D which remained at 3 due to data gaps (i.e., for most outcomes, a lack of: informative human studies, animal studies from multiple species or spanning multiple generations, studies of other organ/system-specific effects associated with other PFAS, including PFOA and PFOS, particularly following developmental exposure). The exception that a UF_S = 1 was applied for all endpoints since no subchronic to chronic extrapolation was required for the candidate toxicity values for the subchronic RfD. For the endocrine endpoint, a UFs = 3 was applied. Although the data are derived from a short-term (28-

day) study, the life cycle of T4 occurs on the order days or less [$t_{½}$ is approximately 12 hours (Döhler et al., 1979)], therefore, the short-term study duration is expected to capture effects for all components of the T4 life cycle and uncertainty about the ability of PFHxA exposure to affect following a 28-day exposure is reduced. Considering this along with the short half-life (hours) of PFHxA in rats (see Section 3.1) and the approximate 3-fold difference in duration between this study and a guideline subchronic study (i.e., 90 days), a $UF_S = 3$ was deemed most appropriate. The resulting candidate toxicity values are shown in Table 5-11.

Table 5-11. Candidate subchronic toxicity values for PFHxA

Endpoint/study/ confidence	Species, strain (sex)	POD_{HED} PFHxA[a] (mg/kg-d)	UF_A	UF_H	UF_S	UF_L	UF_D	UF_C	Candidate value PFHxA (mg/kg-d)	Candidate value PFHxA-Na[c] (mg/kg-d)	Candidate value PFHxA-Na[c] (mg/kg-d)
Hepatic effects											
↑Hepatocellular hypertrophy, 90 d Loveless et al. (2009) *High* confidence	Rat, Crl:CD(SD) (male)	0.11[b]	3	10	1	1	3	100	1×10^{-3}	1×10^{-3}	1×10^{-3}
Hematopoietic effects											
↓Red blood cell, 90 d Chengelis et al. (2009b) *High* confidence	Rat, Crl:CD(SD) (female)	0.078	3	10	1	1	3	100	8×10^{-4}	8×10^{-4}	8×10^{-4}
Developmental effects											
↓Postnatal (F_1) body weight, PND 0 Loveless et al. (2009) *High* confidence	Rat, Sprague-Dawley, F1 (combined)	0.048[b]	3	10	1	1	3	100	5×10^{-4}	5×10^{-4}	5×10^{-4}
↑Perinatal Mortality, PND 0-21 Iwai and Hoberman (2014) High confidence	Mouse, CD-1, F1 (combined)	0.21	3	10	1	1	3	100	2×10^{-3}	2×10^{-3}	2×10^{-3}
Endocrine effects											
↓ Total T4 NTP (2018) *High* confidence	Rat, Harlan Sprague-Dawley (male)	0.29	3	10	3	1	3	300	1×10^{-3}	1×10^{-3}	1×10^{-3}

[a]The RfD for the free acid of PFHxA is calculated using the ratio of molecular weights as described above.
[b]POD_{HED} multiplied by normalization from the sodium salt to free acid (MW free acid/MW sodium salt = 314/336 = 0.935).
[c]To calculate subchronic candidate values, osRfDs or the subchronic RfD for salts of PFHxA, multiply the value of interest by the ratio of molecular weights of the salt and free acid. For example, for the sodium salt of PFHxA, the candidate value is calculated by multiplying the free acid candidate value by 1.070: (MW free acid/MW sodium salt = 336/317 = 1.070).

Selection of Subchronic Organ- or System-Specific RfDs

As described above, subchronic osRfDs associated with each health effect are presented as they may be useful for certain decision purposes (i.e., site-specific risk assessments with less-than-lifetime exposures). Confidence in each subchronic osRfD are described in Table 5-12 and consider confidence in the study used to derive the quantitative estimate, the overall health effect, specific evidence base, and quantitative estimate for each subchronic osRfD.

Table 5-12. Confidence in the subchronic organ/system specific RfDs for PFHxA

Confidence categories	Designation	Discussion
Hepatic subchronic osRfD = 1×10^{-3} mg/kg-d PFHxA; 1×10^{-3} mg/kg-d PFHxA-Na or PFHxA-NH_4		
Confidence in the study used to derive the subchronic osRfD	*High*	Confidence in the study (Loveless et al., 2009) is *high* based on the study evaluation results (i.e., rated high confidence overall) HAWC link) and characteristics that make it suitable for deriving toxicity values, including relevance of the exposure paradigm (route, duration, and exposure levels), use of a relevant species, and the study size and design.
Confidence in the evidence base for hepatic effects	*Medium*	Confidence in the oral toxicity evidence base for hepatic effects is *medium* based on consistent, dose-dependent, and biologically coherent effects on organ weight and histopathology observed in multiple *high* confidence subchronic and chronic studies. The available mechanistic evidence also supports biological plausibility of the observed effects. Limitations of the evidence base for hepatic effects is the lack of human studies, and measures that would have been useful to inform the pathways for hepatic effects leading to hepatocellular hypertrophy (e.g., specific stains for hepatic vacuole contents, specific histological for pathology).
Confidence in quantification of the POD_{HED}	*Medium*	Confidence in the quantification of the POD and subchronic osRfD is *medium* given the POD was based on BMD modeling within the range of the observed data and dosimetric adjustment based on PFHxA-specific PK information. Some residual uncertainty in the application of the dosimetric approach described above is that V_d and CL were not measured in humans or mice and were considered equivalent to those in monkeys and rats, respectively.
Overall confidence in the hepatic subchronic osRfD	*Medium*	The overall confidence in the subchronic osRfD is *medium* and is primarily driven by *medium* confidence in the overall evidence base for hepatic effects, *high* confidence in the study, and *medium* confidence in quantitation of the POD. *High* confidence in the study was not interpreted to warrant changing the overall confidence from medium.
Hematopoietic subchronic osRfD = 8×10^{-4} mg/kg-d PFHxA; 8×10^{-4} mg/kg-d PFHxA-Na or PFHxA-NH_4		
Confidence in study used to derive the subchronic osRfD	*High*	Confidence in the study (Chengelis et al., 2009b) is *high* based on the study evaluation results (i.e., rated high confidence overall) (HAWC link) and characteristics that make it suitable for deriving toxicity values, including relevance of the exposure paradigm (route, duration,

Confidence categories	Designation	Discussion
		and exposure levels), use of a relevant species, and the study size and design.
Confidence in evidence base for hematopoietic effects	*High*	Confidence in the evidence base for hematopoietic effects was *high* based on consistent and biologically coherent effects on red blood cells, hemoglobin, and other hematological parameters measured across multiple *high* confidence chronic and subchronic studies. The RBC and hemoglobin findings were also coherent with an erythrogenic response indicated by increased reticulocytes and pathological findings of splenic extramedullary hematopoiesis and bone marrow erythroid hyperplasia. Limitations of the hematopoietic evidence base are lack of human studies, and some hematological measures were observed only at the highest dose, limiting interpretation of dose-response.
Confidence in quantification of the POD_{HED}	*Low*	Confidence in the quantification of the POD and subchronic osRfD is *low* given the POD was far below the NOAEL (50 mg/kg-d) and the osRfD is far below toxicity values derived for the same finding from other subchronic studies suggesting some underlying variability driving the POD lower.
Confidence in hematopoietic subchronic osRfD	*Medium-Low*	The overall confidence in the subchronic osRfD is *medium-low* and is primarily driven by *low* quantitation of the POD. *High* confidence in the study was not interpreted to warrant changing the overall confidence from *medium-low*.
Developmental subchronic osRfD = 5×10^{-4} mg/kg-d PFHxA; 5×10^{-4} mg/kg-d PFHxA-Na or $PFHxA\text{-}NH_4$		
Confidence in study used to derive the subchronic osRfD	*High*	Confidence in the study (Loveless et al., 2009) is *high* based on the study evaluation results (i.e., rated high confidence overall) (HAWC link) and characteristics that make it suitable for deriving toxicity values, including relevance of the exposure paradigm (route, duration, and exposure levels), use of a relevant species, and the study size and design.
Confidence in evidence base for developmental effects	*Medium*	Confidence in evidence base for developmental effects is *medium* based on the availability of data from two studies in different species (i.e., rats and mice) that consistently observed decreases in offspring body weight and coherent increases in mortality. One area of uncertainty is that there were no multigenerational studies available. Also, data to inform other organ/system-specific hazards (e.g., thyroid, immune, nervous system) following a developmental exposure is lacking. Together these present significant data gaps in the potential effects during this sensitive life stage.
Confidence in the quantification of the POD_{HED}	*Medium*	Confidence in the quantification of the POD and subchronic osRfD is *medium* given the POD was based on BMD modeling, within the range of the observed data and dosimetric adjustment based on PFHxA-specific PK information. Some residual uncertainty in the application of the dosimetric approach described above is that V_d and CL were not measured in humans or mice and were considered equivalent to those in monkeys and rats, respectively.

Confidence categories	Designation	Discussion
Confidence in developmental subchronic osRfD	*Medium*	The overall confidence in the subchronic osRfD is *medium* and is primarily driven by *medium* confidence in the overall evidence base for developmental effects, *high* confidence in the study, and *medium* confidence in quantitation of the POD. *High* confidence in the study was not interpreted to warrant changing the overall confidence from *medium*.
Endocrine subchronic osRfD = 1 × 10^{-3} mg/kg-d PFHxA; 1 × 10^{-3} mg/kg-d PFHxA-Na or PFHxA-NH_4		
Confidence in study used to derive the subchronic osRfD	*High*	Confidence in the study (NTP, 2018) is *high* based on the study evaluation results (i.e., rated high confidence overall) (HAWC link) and characteristics that make it suitable for deriving toxicity values, including relevance of the exposure paradigm (route and exposure levels), use of a relevant species, and the study size and design.
Confidence in evidence base for endocrine effects	*Medium-Low*	Confidence in the oral toxicity evidence base for endocrine effects is *medium-low* based primarily on strong dose-dependent effects on serum T4 in males reported in a *high* confidence short-term study. This finding is consistent with effects observed in a low confidence human study and a zebrafish study. The available mechanistic evidence supports biological plausibility of the observed effects in rodents. Limitations of the evidence base for endocrine effects is the lack of informative human studies and longer duration studies in animals. Additionally, studies to inform the pathways for thyroid effects leading changes in thyroid hormone levels and/or sex specific differences.
Confidence in the quantification of the POD_{HED}	*Medium*	Confidence in the quantification of the POD and subchronic osRfD is *medium* given the POD was based on BMD modeling within the range of the observed data and dosimetric adjustment based on PFHxA-specific PK information. Some residual uncertainty in the application of the dosimetric approach described above is that V_d and CL were not measured in humans or mice and were considered equivalent to those in monkeys and rats, respectively.
Confidence in endocrine subchronic osRfD	*Medium*	The overall confidence in the subchronic osRfD is *medium* and is primarily driven by *medium-low* confidence in the overall evidence base for developmental effects, *high* confidence in the study, and *medium* confidence in quantitation of the POD. *High* confidence in the study was not interpreted to warrant changing the overall confidence from *medium*.

Selection of Subchronic RfD and Confidence Statement

Organ/system-specific subchronic RfD values for PFHxA are summarized in Table 5-13.

Table 5-13. Subchronic osRfD values for PFHxA

System	Basis	POD_{HED}	UF_C	osRfD for PFHxA (mg/kg-d)	osRfD for PFHxA-Na[a] (mg/kg-d)	osRfD for PFHxA-NH_4[a] (mg/kg-d)	Confidence
Hepatic	Increased hepatocellular hypertrophy in adult male Crl:CD Sprague-Dawley rats	0.11 mg/kg-d based on $BMDL_{10ER}$ and free salt normalization (Loveless et al., 2009)	100	1×10^{-3}	1×10^{-3}	1×10^{-3}	*Medium*
Hematopoietic	Decreased red blood cells in adult female Crl:CD Sprague-Dawley rats	0.078 mg/kg-d based on $BMDL_{1SD}$ (Chengelis et al., 2009b)	100	8×10^{-4}	8×10^{-4}	8×10^{-4}	*Medium-Low*
Developmental	Decreased postnatal (PND 0) body weight in F_1 Sprague-Dawley male and female rats, exposed throughout lactation and gestation	0.048 mg/kg-d based on $BMDL_{5RD}$ and free salt normalization (Loveless et al., 2009)	100	5×10^{-4}	5×10^{-4}	5×10^{-4}	*Medium*
Endocrine	Decreased total T4 in adult male Harlan Sprague-Dawley rats	0.29 mg/kg-d based on $BMDL_{1SD}$ (NTP, 2018)	300	1×10^{-3}	1×10^{-3}	1×10^{-3}	*Medium*

[a]To calculate candidate values for salts of PFHxA, multiply the candidate value of interest by the ratio of molecular weights of the free acid and the salt. For example, for the sodium salt of PFHxA, the candidate value would be calculated by multiplying the free acid candidate value by 1.070 (MW free acid/MW sodium salt = 336/314 = 1.070). This same conversion can be applied to other salts of PFHxA, such as the ammonium salt.

From the identified targets of PFHxA toxicity and derived subchronic osRfDs (see Table 5-13), a ***subchronic RfD of 5 × 10^{-4} mg/kg-day based on decreased postnatal body weight*** is selected for less-than-lifetime exposure. Confidence in the subchronic RfD is medium, based on medium confidence in the developmental subchronic RfD, as described in Table 5-12. The confidence in the selected subchronic RfD is equivalent to that of the hepatic subchronic RfDs and higher than the hematopoietic subchronic RfD. The developmental subchronic RfD is expected to be protective of all life stages. The UF_C (see Table 5-13) is lower than or equivalent to the other subchronic osRfDs and the endpoint has the lowest POD_{HED} (0.048 mg/kg-day, see Table 5-11). The

decision to select the developmental subchronic RfD was based on all the available subchronic osRfDs in addition to overall confidence and composite uncertainty for those subchronic osRfDs.

5.2.2. Inhalation Reference Concentration (RfC)

No published studies investigating the inhalation effects of subchronic, chronic, or gestational exposure to PFHxA in humans or animals have been identified. Therefore, an RfC is not derived.

5.3. CANCER TOXICITY VALUES

As discussed in Sections 3.3 and 4.2, given the sparse evidence base and in accordance with the *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005), EPA concluded that there is ***inadequate information to assess carcinogenic potential*** for PFHxA for any route of exposure. Therefore, consistent with the *Guidelines* and the lack of adequate data on the potential carcinogenicity of PFHxA, quantitative estimates for either oral (oral slope factor, OSF) or inhalation (inhalation unit risk; IUR) exposure were not derived.

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