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EPA-SAB-xx-xxx

The Honorable Michael Regan
U.S. Environmental Protection Agency
1200 Pennsylvania Avenue, N.W.
Washington, D.C. 20460

Subject: Transmittal of the Science Advisory Board Report titled, "Review of EPA's Analyses to support EPA's National Primary Drinking Water Rulemaking for PFAS"

Dear Administrator Regan,

Please find enclosed the final report from the Scientific Advisory Board (SAB). The EPA's Office of Water requested that the SAB review four EPA documents titled, *EPA's Proposed Approaches to the Derivation of a Draft Maximum Contaminant Level Goal for Perfluorooctanoic Acid (PFOA) (CASRN 335-67-1) in Drinking Water*; *EPA's Proposed Approaches to the Derivation of a Draft Maximum Contaminant Level Goal for Perfluorooctane Sulfonic Acid (PFOS) (CASRN 1763-23-1) in Drinking Water*; *EPA's Draft Framework for Estimating Noncancer Health Risks Associated with Mixtures of Per- and Polyfluoroalkyl Substances (PFAS)*; and *EPA's Analysis of Cardiovascular Disease Risk Reduction as a Result of Reduced PFOA and PFOS Exposure in Drinking Water*. As part of the proposed rulemaking process, EPA prepared these four documents to support the development of a Maximum Contaminant Level Goal (MCLG) and National Primary Drinking Water Regulation (NPDWR) for per- and polyfluoroalkyl substances (PFAS) under the Safe Drinking Water Act (SDWA).

In response to the EPA's request that the SAB review EPA's four draft documents listed above, the SAB identified subject matter experts to augment the SAB Chemical Assessment Advisory Committee (CAAC) and assembled the SAB PFAS Review Panel to conduct the review. The SAB PFAS Review Panel met virtually via a video meeting platform on December 16, 2021, and then at three (3) subsequent meetings on January 4, 6 and 7, 2022 to deliberate on the agency's charge questions. Another virtual meeting was held on May 3, 2022 and May 6, 2022 to discuss their draft report. Oral and written public comments were considered throughout the advisory process. This report conveys the consensus advice of the SAB.

The SAB recognizes the time constraints for completing the rule-making process and is supportive of the EPA's efforts to utilize the latest scientific finding to inform their decisions. The SAB applauds the Agency's efforts to develop new approaches for assessing the risk of PFAS mixtures and the benefits arising from reducing exposure to these chemicals.

In general, the SAB agreed with many of the conclusions presented in the assessments, framework and analysis. The SAB also identified many areas that would benefit from further clarification to enhance their transparency and increase their utility. While the SAB provided numerous recommendations, we would like to highlight the following ones, with additional details described within the full report. The SAB recommends that EPA consider the following points as they revise their documents:

Draft MCLG documents

- EPA should provide additional transparency and completeness in its evidence identification methodology, including development of a protocol with clear inclusion/exclusion criteria and study evaluation approaches.
- Studies, particularly human studies, that were included in the 2016 health effects summary documents (HESDs) should be considered in the same manner as the more recent studies.
- A consistent framework and descriptors should be used for evidence synthesis and integration for each health outcome.
- In the short-term and in consideration of the Agency's time constraints, EPA should initially focus on those health outcomes that have been concluded to have the strongest evidence, including the liver, immune system, serum lipids, and fetal growth.
- The process of hazard identification be separated from the process of dose-response assessment.
- The rationale and criteria for selection of endpoints and specific studies for point of departure (POD) development should be more clearly presented.
- EPA should use ALT as an endpoint in light of the numerous studies in the literature that support an association between slight elevations in ALT and increased risk of morbidity and/or mortality.
- While the Panel agrees with the "likely" designation for PFOA carcinogenicity based on new evidence and prior evidence included in the 2016 HESD, a more structured and transparent "weight of evidence" discussion to support the rationale behind this designation is needed.
- EPA should develop multiple candidate cancer slope factors (CSFs), including those based on additional epidemiologic studies of sufficient quality, as well as animal cancer bioassays. Strengths and limitations for each study should be discussed and a judgment made as to whether to select one or more studies to represent the overall slope factor.
- EPA needs to provide additional details and transparency for all quantitative modeling, including that used for CSF development, toxicokinetic modeling, and benchmark dose modeling for POD derivation. It is essential that details of the BMD modeling that forms the basis of the PODs is transparently available for evaluation of the methods, approaches, and results.

- EPA should reconsider its choice of the Verner *et al.* (2016) human toxicokinetic model and consider whether the Goeden *et al.* (2019) model, which incorporates age-specific toxicokinetic and exposure factors in predicting internal dose, is more appropriate for use in the development of the PFOA and PFOS reference doses (RfDs) and MCLGs.
- For the development of a non-cancer RfD, EPA should consider multiple human and animal studies for a variety of endpoints in different populations so as to provide convergent evidence that is more reliable than any single study or health endpoint in isolation.
- EPA should provide a stronger and more transparent justification for the choice of benchmark responses (BMRs) - not only for decreased antibody response, but also other endpoints for which BMDs were developed.
- In deriving RfDs, EPA should consider using an “internal dose” POD, which is then further adjusted for inter-species and intra-species uncertainty/variability, so that the RfD is expressed in a dose metric equivalent value. This internal dose RfD can then be converted using toxicokinetic (TK) modeling to either an equivalent external dose or an equivalent water concentration, as appropriate.
- The Panel agrees with the uncertainty factors (UFs) used in deriving RfDs, but EPA should consider adoption of a probabilistic framework to calculate risk-specific doses and evaluate potential methods for accounting for mixtures in establishing MCLGs.
- The Panel supports the selection of an RSC of 20%, but EPA should revise certain aspects of the RSC sections in the draft MCLG documents to better describe and explain the rationale for arriving at an RSC of 20%.

Draft mixtures document

- The SAB PFAS Review Panel supports dose additivity based on a common outcome, instead of a common mode of action as a health protective default assumption and does not propose another default approach. However, EPA should more thoroughly and clearly presented the uncertainties associated with this approach along with information supporting this approach.
- The Panel expressed concern regarding the requirement for “external peer review” of toxicity values developed by states and recommends that this phrase in the draft framework be broadened to recommend the need for scientific input and review in general.
- EPA should consider using a menu-based framework to support selection of fit-for-purpose approaches, rather than a tiered approach as described in the draft Mixtures document. Tiered approaches that require increasingly complex information before reaching a final decision point can be extremely challenging for data-poor chemicals such as PFAS.
- EPA should provide clarification regarding the conceptual similarities and differences between the target-organ-specific hazard index (TOSHI) approach, the relative potency factor (RPF) approach, and the mixture benchmark dose (BMD) approach, since all are based on health effect-specific values (i.e., Reference Values (RfVs) or RPFs) for the individual PFAS in the PFAS mixture. More discussion and comparison of approaches, as well as when they converge, is needed. For instance, given the mathematical

correspondence between the RPF and mixture BMD approaches, EPA should consider revising the discussion of these two approaches to present them as essentially the same (or highlighting any essential differences), and perhaps also merging them into a single section.

- For both the RPF and mixture BMD approach, EPA's approach would be strengthened by the use of PODs from animal studies that are based on human equivalent doses (HEDs) rather than administered doses. The Panel found it difficult to envision situations in which the mixture BMD was advantageous, so EPA should provide additional information on how the proposed Mixtures BMD approach will be applied in practice.

CVD document

- The Panel supports the overall approach to estimating reductions in cardiovascular disease (CVD) risk associated with reductions in exposure to PFOA and PFOS in drinking water. However, the Panel expressed a concern about the apparent discrepancy between the draft CVD document's focus on CVD risk, and the draft MCLG documents' conclusions that the evidence of CVD was not sufficient to form the basis of a RfD. EPA should provide more discussion as to the rationale for selecting this particular endpoint for risk reduction analysis, as well as consider risk reduction analyses for other endpoints.
- EPA should ensure that recommendations for the draft MCLG documents relating to evidence identification and synthesis are applied to the CVD endpoint.
- The Panel generally agrees with the meta-analysis, life-table and risk estimation methods, but EPA should provide additional clarity as to the application of these approaches, including conduct of sensitivity analyses.
- While the ASCVD model is a reasonable choice for estimating the probability of first time CVD events, it is not without limitations. EPA should include more discussion of the accuracy of its predictions, particularly for sub-populations.
- EPA should evaluate whether inclusion of HDL data would influence the results.

As the EPA finalizes the draft MCLG, Mixtures and CVD documents, the SAB encourages the EPA to address the SAB's concerns raised in the enclosed report and consider their advice and recommendations. The SAB appreciates this opportunity to review EPA's Analyses to support EPA's National Primary Drinking Water Rulemaking for PFAS and looks forward to the EPA's response to these recommendations.

Sincerely,

Alison Cullen, Ph.D.
Chair
EPA Science Advisory Board

Weihsueh Chiu, Ph.D.
Chair
EPA SAB PFAS Review Panel

Enclosure

NOTICE

This report has been written as part of the activities of the EPA Science Advisory Board, a public advisory committee providing extramural scientific information and advice to the Administrator and other officials of the Environmental Protection Agency. The Board is structured to provide balanced, expert assessment of scientific matters related to problems facing the Agency. This report has not been reviewed for approval by the Agency and, hence, the contents of this report do not represent the views and policies of the Environmental Protection Agency, nor of other agencies in the Executive Branch of the Federal government, nor does mention of trade names or commercial products constitute a recommendation for use. Reports of the EPA Science Advisory Board are posted on the EPA website at <http://www.epa.gov/sab>.

Science Advisory Board (SAB) Draft Report (April 1, 2022) to Assist Meeting Deliberations -- Do Not Cite or Quote --This draft is a work in progress, does not reflect consensus advice or recommendations, has not been reviewed or approved by the chartered SAB and does not represent EPA policy.

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ACRONYMS AND ABBREVIATIONS

ADME	Absorption, Distribution, Metabolism, Excretion
ALT	Alanine Aminotransferase
APFO	Ammonium Perfluorooctanoate
ASCVD	Atherosclerotic Cardiovascular Disease
BMD	Benchmark Dose
BMDL	Benchmark Dose lower bound limit
BMR	Benchmark Response
CSF	Cancer Slope Factor
DA	Dose Additivity
EFSA	European Food Safety Authority
GCA	General Concentration Addition
Gen-X	dimeracid and HFPO dimer acid ammonium salt
HAWC	Health Assessment Workspace Collaborative
HBWC	Health-Based Water Concentrations
HDL	High-Density Lipoprotein
HDLC	High-Density Lipoprotein Cholesterol
HESD	Health Effects Support Document
HED	Human Equivalent Dose
HFPO	hexafluoropropylene oxide
HI	Hazard Index
IC	Index Chemical
IRIS	Integrated Risk Information System
LOAEL	Lowest Observed Adverse Effect Level
OR	Odds Ratio
PECO	Population (including animal species), Exposure, Comparator, and Outcomes
PFAS	Per- and Polyfluoroalkyl Substances
PFBA	Perfluorobutanoic Acid
PFBS	perfluorobutane sulfonic acid
PFOA	Perfluorooctanoic Acid
PFOS	Perfluorooctane Sulfonic Acid
PFCA	Perfluorocarboxylic Acid
PFSA	Perfluorosulfonic Acid
POD	Point of Departure
PPAR-a	Peroxisome Proliferator-Activated Receptor - alpha
PWS	Public Water System
MCLG	Maximum Contaminant Level Goal
MCL	Maximum Contaminant Level
MOA	Mode of Action/Mechanism of Action
NAM	New Alternative Methods
NASEM	National Academies of Sciences, Engineering, and Medicine
NPDWR	National Primary Drinking Water Regulations
RCC	Renal Cell Carcinoma

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1	RfD	Reference Dose
2	RfV	Reference Values
3	RPF	Relative Potency Factor
4	RSC	Relative Source Contribution
5	SDWA	Safe Drinking Water Act
6	TC	Total Cholesterol
7	TOSHI	Target-Organ-Specific Hazard Index
8	UF	Uncertainty Factor

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INTRODUCTION

The U.S. Environmental Protection Agency (EPA), having initiated the process to develop a Maximum Contaminant Level Goal (MCLG) and National Primary Drinking Water Regulation (NPDWR) for per- and polyfluoroalkyl substances (PFAS) under the Safe Drinking Water Act (SDWA), is seeking comment from the EPA Science Advisory Board (SAB) on key scientific issues related to the development of the NPDWR. As part of this proposed rulemaking, EPA has prepared the following four documents:

1. *EPA's Proposed Approaches to the Derivation of a Draft Maximum Contaminant Level Goal for Perfluorooctanoic Acid (PFOA) (CASRN 335-67-1) in Drinking Water*
2. *EPA's Proposed Approaches to the Derivation of a Draft Maximum Contaminant Level Goal for Perfluorooctane Sulfonic Acid (PFOS) (CASRN 1763-23-1) in Drinking Water*
3. *EPA's Draft Framework for Estimating Noncancer Health Risks Associated with Mixtures of Per- and Polyfluoroalkyl Substances (PFAS)*
4. *EPA's Analysis of Cardiovascular Disease Risk Reduction as a Result of Reduced PFOA and PFOS Exposure in Drinking Water.*

The draft *EPA's Proposed Approaches to the Derivation of a Draft Maximum Contaminant Level Goal for PFOA and PFOS* documents (henceforth referred to as “the draft MCLG documents”) provide a number of values, including toxicity values, that could be use in a human health risk assessment. The EPA stated that these documents do not constitute risk assessments, however the values presented were derived using human health risk assessment guidance, guidelines, and current methods.

The draft *EPA's Draft Framework for Estimating Noncancer Health Risks Associated with Mixtures of Per- and Polyfluoroalkyl Substances (PFAS)* document (henceforth referred to as “the draft mixtures document”) provides a draft approach for a tiered, data-driven framework for estimating the likelihood of noncancer human health risks associated with oral exposures to mixtures of PFAS, based on EPA guidelines and guidance (U.S. EPA, 1986, 2000). The EPA maintains that although the framework and illustrative examples provided in this document include examples for PFAS in water, the framework itself is not limited to specific media and may be useful for understanding the potential non-cancer health effects of PFAS mixtures under various authorities or decision contexts. The EPA contends that the approach presented is not intended to be used to assign groups or subclasses or otherwise classify PFAS. Rather, the framework is designed for practical application of EPA chemical mixtures approaches and methods for a particular exposure to gain insight on the potential health risk(s) associated with exposure to mixtures of PFAS. The EPA notes that the draft mixtures document with the associated illustrative examples is intended to inform PFAS evaluation(s) by federal, state, and tribal partners, as well as public health experts, drinking water utility personnel, and other stakeholders interested in assessing the potential human health hazards and risks associated with PFAS mixtures. The EPA further emphasized that the draft mixtures document is not a

regulation and does not impose legally binding requirements on EPA, states, tribes, or the regulated community, and might not apply to a particular situation based on the circumstances.

EPA stated that it is considering several adverse health effects for quantified health risk reduction assessment. Among the adverse health effects that EPA found to have sufficient weight of evidence and available data to inform estimates of avoided adverse health outcomes were the effects of PFOA and PFOS on serum lipids, specifically total cholesterol (TC), a well-established risk factor for cardiovascular disease. EPA's *Analysis of Cardiovascular Disease Risk Reduction as a Result of Reduced PFOA and PFOS Exposure in Drinking Water* document (henceforth referred to as "the CVD document") presents EPA's methodology to determine the avoided cases of CVD events (e.g., first heart attack, first stroke, death from coronary heart disease) for a hypothetical public water system (PWS).

The EPA's Office of Water requested that the SAB review EPA's four draft documents listed above. In response to the EPA's request, the SAB identified subject matter experts to augment the Science Advisory Board (SAB) Chemical Assessment Advisory Committee (CAAC) and assembled the SAB PFAS Review Panel to conduct the review. The SAB PFAS Review Panel met virtually using a video meeting platform on December 16, 2021, and then at three (3) subsequent meetings on January 4, 6 and 7, 2022 to deliberate on the agency's charge questions. Another virtual meeting was held on May 3, 2022 and May 6, 2022 to discuss the Panel's draft report. Consideration of oral and written public comments was encouraged throughout the advisory process.

The Panel identified numerous instances in which the analyses and approaches in EPA's documents could be revised to be more thorough and transparent. This report is organized into three sections, one for each topic area (i.e., MCLG derivation, Mixtures approaches, and Benefits from CVD reduction). In each section, the charge questions raised by the agency are presented and then followed by the consensus response and recommendations. The Panel provided key recommendations that are necessary to improve the critical scientific concepts, issues, and/or narrative within the EPA's documents. The Panel deemed these recommendations as important for improving the transparency of the agency's conclusions and to bolster the supporting evidence for them.

A list of acronyms and abbreviations can be found at the front of this report to assist in orienting the reader to the terminology used in the EPA's documents and throughout the Panel's responses to the Charge Questions. Editorial comments are presented within Appendix A. All materials and comments related to this report are available at:

https://sab.epa.gov/ords/sab/f?p=114:18:31389248712411:::RP,18:P18_ID:2601

SECTION I - MCLG derivation

Proposed Approaches to the Derivation of a Draft Maximum Contaminant Level Goals for PFOA and PFOS in Drinking Water

Charge Question #1- Study Identification and Inclusion

EPA used systematic review methods consistent with the current ORD systematic review practice to ensure transparency and completeness of literature identification, sorting, and study quality evaluation. Is the process clearly described? Please identify additional peer-reviewed studies that the panel is aware of that could inform toxicity value derivation.

Overall Comments

While the Panel agreed that some aspects of the process were clearly described (e.g., the literature search and screening processes), they also identified multiple inconsistencies and deficiencies in both the description and execution of the systematic review process utilized in the evaluation of both PFOA and PFOS. The following sections, organized by steps in the systematic review process, offer a short background on the step, provide feedback from the Panel on the step, and close with recommendations for moving forward.

Problem formulation and protocol development

Background

Before beginning a systematic review process, it is essential to clearly define the study question to be addressed and to develop a protocol. Often a preliminary literature review is conducted, which *a priori* describes the approach and criteria that will be used to guide the review. A systematic review protocol makes both the process and the methods of the review transparent and offers an opportunity for the peer review of methods. A clearly defined and published protocol creates a public record of the steps that will be used to conduct each step of the review – including evidence identification, evaluation, and synthesis.

Review of EPA process

The Panel and the public commenters noted significant concerns that the reviews for PFOA and PFOS do not appear to have established a predefined protocol. The lack of a protocol led to a lack of clarity across each of the major systematic review steps for both chemicals and was seen as a major deficiency of the reviews.

Recommendations

Although it is not possible at this point to establish a protocol for the existing review process, the Panel recommends that EPA add additional clarification and correction to the existing systematic reviews to fill in gaps about how specific tasks were completed. Furthermore, when designing additional reviews for sensitive endpoints identified as having the strongest evidence in the draft MCLG documents (see Panel recommendations for charge question #2- noncancer hazard

identification), EPA should establish protocols prior to beginning any new systematic review process for these endpoints. Protocol development will not only help increase transparency of the subsequent reviews but may also aide in coordinating multiple teams that are working across various endpoints.

Evidence Identification

Background

During the evidence identification step of a systematic review, evidence related to the study section is sought and selected based upon eligibility criteria from the Population (including animal species), Exposure, Comparator, and Outcomes (PECO) statement. Searching for and selecting evidence should be conducted using a detailed search plan that includes databases to be searched, gray literature sources, and other search methods including whether existing systematic reviews will be utilized. Literature searches can yield large numbers of records and are usually carried out in a tiered approach – beginning with a title and abstract search followed by a full text review. A range of software tools can be used in the management of evidence review and selection.

Review of EPA process

In general, the Panel found the literature search and screening processes clearly described. The Panel noted that the use of SWIFT and DistillerSR to sort the literature and process for quality evaluation and confidence determination were reasonable steps. However, several areas of concern were noted.

The first area of concern was that the PECO statement was deficient in the list of “relevant forms” of PFOA and PFOS due to a lack of inclusion of relevant salts (e.g., salts of PFOA such as ammonium perfluorooctanoate (APFO), sodium perfluorooctanoate, and potassium perfluorooctanoate, or salts of PFOS such as lithium perfluorooctanesulfonate, potassium perfluorooctanesulfonate (K+PFOS), ammonium perfluorooctanesulfonate, or sodium perfluorooctanesulfonate). The omission of salts of PFOA and PFOS from PECO criteria for human and animal studies is problematic because a salt of PFOA (e.g., APFO) or PFOS (e.g., K+PFOS) was used as the test compound in all of the toxicity studies of PFOA and PFOS in laboratory animals reviewed in the draft MCLG documents, and some studies of occupationally exposed workers also use these terms. These salts completely dissociate to the anionic form (PFOA, PFOS) in the body, in water, and in other environmental media. While it is likely that the publications reporting the animal toxicity studies include the term "PFOA" or "PFOS" as well the salt of PFOA or PFOS that was used, the salts should be included as "relevant forms" in the PECO criteria.

In addition to the lack of inclusion of salts in the PECO statement, the Panel found that the exact process used to inform inclusion and exclusion of evidence was unclear. They found that the lack of documentation about excluded literature, particularly for studies that underwent full text review, made an evaluation of whether inclusion/exclusion criteria were followed challenging. For example, the Panel found that Vriens *et al.* (2019) was not listed in the PFOA or PFOS

documents despite it being a study that could inform mechanisms. In addition to missing mechanistic studies, the Panel also found that the inclusion and exclusion of epidemiologic and animal studies were inconsistent across endpoints, leading to confusion about the criteria being used. For example, Table 2 (p.11) of the PFOA document states: "*Studies with less than 28 days of dosing, with the exception of reproductive or developmental studies, should be tagged as supplemental,*" but Section 3.1.3 (p.24), which is about literature search results for animal toxicology studies, states: "*of the 32 animal studies that met the inclusion criteria, most studies had either short-term (n = 15) or developmental (n = 12) study designs.*" The inclusion of short-term studies other than developmental studies in Section 3.1.3 does not seem to be consistent with the information in Table 2.

In addition to the lack of clarity about inclusion and exclusion criteria, the Panel concluded that the decision to exclude literature published within the timeframe of the development of the 2016 health effects support document (HESD; U.S.EPA, 2016) in the current literature search was unjustified. The 2016 HESD did not include systematic reviews of the literature and are not fully representative of the relevant literature needed for the PFOA and PFOS reviews. For example, a 2011 study by Dong *et al.* which reported that PFOS caused a dose-related decrease in serum levels of sheep red blood cells (SRBC)-specific IgM in mice was not included in the 2016 HESD. Dong *et al.* (2011) is important because it was used as the critical study for the PFOS Reference Doses (RfDs) and drinking water guidelines established by Minnesota, New Hampshire, and Washington (Post, 2021). The lack of inclusion of this key study in the 2016 HESD document erodes confidence that all the necessary and relevant studies needed for the current PFOA and PFOS reviews are included.

Another important issue identified was that the rationale for not considering studies, particularly human studies, that were included in the 2016 HESDs is not clear or supportable. There is no reason to conclude that the earlier studies are less relevant or of lesser quality than the newer studies. Consideration of all human studies is especially important because conclusions about the level of evidence for human health effects, which are generally observational rather than experimental, are based on the overall weight of evidence from all relevant data. Furthermore, specific human epidemiology studies included in the 2016 HESDs may be preferable to the more recent studies for point of departure (POD) development for some health endpoints, as illustrated by the use of data from Grandjean *et al.* (2012) as the basis for the final RfDs for both PFOA and PFOS. This topic is discussed in more detail, including examples, in the response to the charge question #2 on Noncancer Hazard Identification.

Finally, the Panel noted that relevant literature has been published since the exclusion date used for the draft MCLG documents. While selecting a date for the search exclusion is a critical element of a systemic review literature search, a cut-off date of greater than a year ago, creates a significant window in which relevant literature may have been published.

Recommendations

The Panel recommends several changes to the evidence identification step of the PFOA and PFOS systematic reviews.

- First, inclusion and exclusion criteria need to be more clearly described.
- Second, a list of excluded evidence after the full-text review should be developed and made publicly accessible. This may help provide clarity about why specific studies were excluded.
- Third, earlier literature from before the search used for the 2016 HESDs must be included in the literature search and considered for both strength of evidence evaluation and dose-response.
- Fourth, the PECO statements should be updated to include salts of PFOA and PFOS.
- Finally, as is noted in the draft ORD handbook (U.S. EPA, 2020), the literature search should be updated, with an established protocol, throughout the draft development such that the full literature search update is less than one year from the final review.

Evidence Evaluation

Background

The evidence evaluation step of a systematic review is the stage at which the individual studies selected for inclusion in the review are assessed using predefined criteria for internal validity (also known as “risk of bias”), study sensitivity (i.e., whether there are factors in a study’s design or conduct that reduce its ability to observe an effect), and study quality (i.e., whether a study is conducted at the highest possible standard for the study type). The goal of this stage of a systematic review is to evaluate whether the results of a study represent a “reliable, sensitive, and informative presentation of a true response”, as described in the draft ORD handbook (U.S. EPA, 2020).

Review of EPA process

In general, the Panel agreed that the evidence evaluation process was difficult to follow and not clearly described. The seven domains that were considered are not listed in the text, and critical details about how the seven domains were selected are missing from the draft MCLG documents, as are details about whether objective criteria were used in weighing how deficiencies in certain/multiple domains contributed to the overall rating. It is unclear whether the EPA used protocols in the 2020 draft ORD handbook for their evaluation steps, a modified version of the protocol, or a completely different process.

The Panel was also concerned about the downgrading of studies based upon “study sensitivity” if they had limited exposure contrasts and/or small sample sizes. Narrow exposure ranges should not automatically lead to downgrading of studies as these can still contribute informative data within that narrow range, despite reductions in precision of effect estimates. Additionally, the reports mention exclusion of publications with specific population subsets. More transparency and information are needed on how this decision was reached and used as a basis for exclusion.

The Panel also noted that a protocol for risk of bias assessment and, more importantly, how that approach was used in the synthesis of evidence for each particular health endpoint is not clearly presented to be confidently judged for accuracy or transparency, or for consistency across health endpoints. This is especially important when a proposed systematic review protocol has not been previously registered or published.

1 **Recommendations**

2 The Panel recommends that the EPA clearly explain the protocols used in its evidence evaluation
3 process. It is critically important to clearly define how each domain in the evaluation protocol is
4 used and to ensure that terms (e.g., “study quality”, “study validity”, and “study risk of bias”) are
5 defined and used consistently.

6 To enhance the transparency of the study evaluations, the Panel recommends that the domains
7 evaluated should be identified in the draft MCLG documents. While they are available in the
8 Health Assessment Workspace Collaborative (HAWC) database, they are not easily located
9 which hinders the ability of readers to review this information.

10 **Data Extraction**

11 *Background*

12 According to the draft 2020 ORD handbook data extraction is the stage of the systematic review
13 during which relevant results from each study are extracted and reviewed to facilitate the
14 drawing of comparisons across results. The results of this process are tables, graphs, and other
15 visualizations that help organize and display study findings.

16 *Review of EPA process*

17 Several issues with the data extraction process used by the EPA were identified. For example,
18 information on where the absorption, distribution, metabolism, and excretion (ADME) and
19 mechanistic data that were extracted can be found does not appear to be included in the draft
20 MCLG documents.

21 Another example of the lack of clarity in data extraction was observed in the Dose-response
22 Studies subsection of the Data Extraction section in the draft MCLG documents (p.19), where
23 the Agency states: “Data extraction was conducted for most studies that were included in the
24 literature inventory, except those excluded as described below. ... Extractions were limited to
25 outcomes of interest and/or the most sensitive LOAEL.” It was not clear whether this means the
26 most sensitive LOAEL was based on internal dose (serum PFOA or PFOS level) or administered
27 dose. For most human studies, the doses are based on serum levels (internal dose). For animal
28 studies, the LOAEL based on administered dose may not correspond to the LOAEL based on
29 internal dose (serum level) because of toxicokinetic (half-life) differences between species or
30 sexes (e.g., much quicker excretion rate in female vs. male rats). The Panel recognizes that
31 evaluating this issue for all studies for which it is relevant may not be possible due to resource
32 limitations. However, the Panel suggests that this potential uncertainty in the approach be
33 acknowledged.

34 Additionally, the Dose-Response Studies subsection of the Data Extraction section (p. 19) where
35 the Agency states that: “...low confidence studies when medium and high confidence studies
36 (e.g., on an outcome) were available” did not undergo data extraction.” This appears to
37 contradict the statement in Section 2.4 that low confidence animal studies were not considered

and that "all study designs" (not specifying study confidence level) were considered for human studies. This information should be clarified and should be consistent in the two sections.

Also, the draft MCLG documents (p.19) state that: *"For human evidence, all study designs were considered; for animal evidence, only animal studies with at least two exposure groups and with high or medium for study quality were considered."* If all human studies were considered, the rationale for considering human studies, but not animal studies, that have low confidence should be provided.

Finally, the critical deficiency that mechanistic data are not summarized in the draft MCLG documents, except for summary tables of the number of studies with each type of mechanistic information, was also observed in the draft MCLG documents. Discussion of the mode(s) and/or mechanism(s) of action for toxicity is normally an important component of toxicity assessments such as these PFOA and PFOS assessments. As discussed in the draft 2020 ORD staff handbook, such an evaluation can provide information about the human relevance of effects observed in animal species and the plausibility of effects observed in humans, among other areas of potential uncertainty. EPA did not provide a rationale for concluding that mechanistic data does not need to be considered in cancer and non-cancer hazard identification and that this information can be added later. While the Panel recognizes that it may not be possible to include an evaluation of mechanistic data for all health effects, a mechanistic or mode of action evaluation for the noncancer endpoint(s) selected as critical endpoints for RfD(s) and for the weight of evidence for carcinogenicity should be included to provide support for EPA's conclusions.

Recommendations

The Panel recommends that the EPA clearly and transparently articulate the processes and final products of data extraction efforts as they revise the draft MCLG documents. If data extracted are not publicly available, this should also be stated in the revised documentation.

The Panel recommends that EPA include mechanistic evaluations for key non-cancer and cancer weight of evidence evaluations.

Evidence synthesis

Background

Evidence synthesis is the stage of a systematic review when the results from individual studies are quantitatively and/or qualitatively analyzed within a specific stream of evidence. The methods that will be used for evidence synthesis should be predetermined in the systematic review protocol to ensure transparency and consistency across all evidence streams for each health outcome evaluated.

Review of EPA process

During the public meeting on December 16, 2021, the EPA noted that the evidence synthesis processes did not utilize a systematic review protocol in the draft MCLG documents.

1 Additionally, the Health Effects Evidence Synthesis and Integration sections for each health
2 outcome in the Hazard Identification Section 3.3 do not appear to consistently follow the process
3 presented in the Systematic Review Section 2.6. Specifically, it is stated on page 22 of the draft
4 MCLG documents that: *"a summary discussion that addresses considerations regarding*
5 *causation as adapted from Hill (1965)"* is provided for each health outcome, but this was not
6 consistently done in the Evidence Synthesis and Integration sections in Hazard Identification
7 (Section 3.3). The Panel noted that this represents a significant deviation from standard
8 Systematic Review protocols, including the process established in the draft 2020 ORD
9 handbook. In general, the Panel found the process used by the agency to be lacking in clarity and
10 transparency. For example, it was noted that it was unclear who synthesized the evidence, how
11 disagreements were resolved, how conflicting results of different studies were accommodated,
12 and whether syntheses were independently reviewed by scientists knowledgeable about the
13 studies.

14 In addition to the lack of a systematic protocol for evidence synthesis, the lack of clarity was
15 further exacerbated by the use of non-integrated teams across the various evidence streams. EPA
16 stated during the Panel meeting on December 16, 2021, that different sections were written by
17 different scientists who used professional judgement as to the terms used and the way
18 conclusions were presented. The inconsistencies introduced due to a disjointed process instead of
19 a structured and systematic approach make it difficult to compare the conclusions of the different
20 health effects sections.

21 The Panel also found the use of inconsistent language confusing and in need of standardization.
22 For example, the conclusions in different Evidence Integration sections use a range of terms that
23 appear to have similar meanings - *"suggestive evidence," "moderate evidence,"* and *"consistent*
24 *evidence"*; *"inconsistent evidence"* and *"mixed evidence"* - but no definitions of these terms were
25 provided. Furthermore, the overall nature of the conclusions for different health effects was not
26 presented consistently (e.g., *"suggestive evidence for an association of PFOS with [the health*
27 *outcome]"*, *"suggestive evidence that PFOA impacts [the health outcome]"*, or *"suggestive*
28 *evidence for an effect of PFOA on [the health outcome]"*). The Panel was unclear from the
29 wording of these conclusions whether they were intended to apply to the evidence for association
30 of the effect with PFOA or to the overall evidence that PFOA causes the effect - an important
31 distinction.

32 Additionally, it is stated on p.22 of both the PFOA and PFOS documents that: *"The syntheses of*
33 *human and animal health effects evidence focused on describing aspects of the evidence that best*
34 *inform causal interpretations, including the exposure context examined in the sets of studies."*
35 The meaning of "exposure context" here is unclear and should be clarified.

36 Finally, the Panel also noted that a "tiered" approach (also referred to as sensitivity analyses)
37 should be considered to evaluate whether the interpretations or conclusions changed based on
38 varied decisions about inclusion or exclusion and a rating of high, medium and low confidence
39 studies across various study design domains.

Recommendations

The Panel suggests that a structured, consistent process and consistent terminology be used for analysis and synthesis of animal evidence, human evidence, and overall evidence be used. One example of such an approach is presented in Chapters 9 and 11 of the draft ORD staff handbook for developing IRIS assessments (U.S. EPA, 2020), and an example of the application of this approach can be found in Sections 3.2 and 4.1 of the draft EPA IRIS assessment of perfluorobutanoic acid (PFBA) (U.S. EPA, 2021). Another approach that could be utilized is that of the National Toxicology Program (NTP) Office of Health Assessment and Translation (OHAT). OHAT also recommends a tier system to characterize the overall risk of bias for each study as a way of comparing the internal validity across the evidence base (NTP OHAT, 2015). As stated earlier, this may not be possible for all health outcomes included in the draft document due to resource limitations. If this is the case, a structured approach should be used to evaluate the evidence for those endpoints that have been concluded to have the strongest evidence.

Additional peer-reviewed studies that could inform hazard identification and toxicity value derivation.

The Panel identified the following key study of immunotoxicity that may be useful:

Dong GH, Liu MM, Wang D, Jin YH, Zheng L, Liang ZF. 2011. Sub-Chronic Effect of Perfluorooctanesulfonate (PFOS) on the Balance of Type 1 And Type 2 Cytokine in Adult C57BL6 Mice. *Arch Toxicol* 85, 1235–1244. <https://doi.org/10.1007/s00204-011-0661-x>

The Panel also identified the following additional epidemiology studies on associations of PFAS and breastfeeding issues:

Nielsen C, Li Y, Lewandowski M, Fletcher T, Jakobsson K, 2022. Breastfeeding Initiation and Duration After High Exposure to Perfluoroalkyl Substances Through Contaminated Drinking Water: A Cohort Study from Ronneby, Sweden, *Environmental Research*, Volume 207, 112206, ISSN 0013-9351, <https://doi.org/10.1016/j.envres.2021.112206>.

Timmermann CAG, Andersen MS, Budtz-Jørgensen E, Boye H, Nielsen F, Jensen RC, Bruun S, Husby S, Grandjean P, Jensen TK, 2022. Pregnancy Exposure to Perfluoroalkyl Substances and Associations with Prolactin Concentrations and Breastfeeding in the Odense Child Cohort, *The Journal of Clinical Endocrinology & Metabolism*, Volume 107, Issue 2, Pages e631–e642, <https://doi.org/10.1210/clinem/dgab638>

The Panel identified the following additional epidemiology studies on associations of PFAS with infectious disease:

Timmermann CAG, Jensen KJ, Nielsen F, Budtz-Jorgensen, van der Klis F, Benn CS, Grandjean P, Fisker AB. 2020. Serum Perfluoroalkyl Substances, Vaccine Responses, and Morbidity in a Cohort of Guinea-Bissau Children. *Environmental Health Perspectives*. 128(8):87002.

Dalsager L, Christensen N, Halekoh U, Timmermann CAG, Nielsen F, Kyhl HB, Husby S, Grandjean P, Jensen TK, Andersen HR. 2021. Exposure To Perfluoroalkyl Substances During Fetal Life and Hospitalization for Infectious Disease in Childhood: A Study Among 1,503 Children from the Odense Child Cohort. *Environ Int*. 149:106395. doi: 10.1016/j.envint.2021.106395. Epub 2021 Jan 25. PMID: 33508532

Bulka CM, Avula V, Fry RC. 2021. Associations of Exposure to Perfluoroalkyl Substances Individually and in Mixtures with Persistent Infections : Recent Findings From NHANES 1999–2016, *Environmental Pollution*, Volume 275, 116619, ISSN 0269-7491. <https://doi.org/10.1016/j.envpol.2021.116619>.

The Panel identified the following additional epidemiology studies on associations of PFAS with bone health:

Buckley JP, Kuiper JR, Lanphear BP, Calafat AM, Cecil KM, Chen A, Xu Y, Yolton K, Kalkwarf HJ, Braun JM, 2021. Associations of Maternal Serum Perfluoroalkyl Substances Concentrations with Early Adolescent Bone Mineral Content and Density: The Health Outcomes and Measures of the Environment (HOME) Study. *Environmental Health Perspectives*, 129(9):097011-1

Banjabi AA, Li AJ, Kumosani TA, Yousef JM, Kannan K. 2020. Serum Concentrations of Perfluoroalkyl Substances and Their Association with Osteoporosis in a Population in Jeddah, Saudi Arabia. *Environ Res*. 187:109676. doi: 10.1016/j.envres.2020.109676. Epub 2020 May 16. PMID: 32485360.

The Panel identified the following studies on PFAS exposure and reduced vaccine response:

Shih YH, Blomberg AJ, Bind MA, Holm D, Nielsen F, Heilmann C, Weihe P, Grandjean P, 2021. Serum Vaccine Antibody Concentrations in Adults Exposed to Per- and Polyfluoroalkyl Substances: A Birth Cohort in The Faroe Islands. *Journal of Immunotoxicology*, 18(1):85-92 (Hepatitis A antibody) <https://pubmed.ncbi.nlm.nih.gov/34143710/>

Timmermann CAG, Pedersen HS, Weihe P, Bjerregaard P, Nielsen F, Heilmann C, Grandjean P. 2022. Concentrations of Tetanus and Diphtheria Antibodies in Vaccinated Greenlandic Children Aged 7-12 Years Exposed to Marine Pollutants, A Cross Sectional Study. *Environmental Research*. 203:111712. (Cross-sectional in Greenlandic children at ages 7-12 years) <https://pubmed.ncbi.nlm.nih.gov/34343554/>

von Holst H, Nayak P, Dembek Z, Buehler S, Echeverria D, Fallacara D, John L. 2021. Perfluoroalkyl Substances Exposure and Immunity, Allergic Response, Infection, and Asthma in Children: Review of Epidemiologic Studies. *Heliyon*, 7:e08160 (review article) <https://pubmed.ncbi.nlm.nih.gov/34712855>

Charge Question #2A - Noncancer Hazard Identification

Please comment on the health effect/outcome categories identified from the review of the available literature. Do you agree with the strong vs. suggestive evidence designations for the various health outcome categories? Do any other health systems or endpoints need to be considered for POD derivation?

General Comments

The charge question had requested input on the “strong” and “suggestive” designations. EPA clarified that they are seeking input on whether the evidence for the endpoints for which PODs were developed is strong enough for dose-response, and if there is strong enough evidence to derive PODs for additional endpoints. Some of the issues addressed here overlap with the responses to the earlier Systematic Review charge questions.

Recognizing that the EPA PFOA and PFOS assessments will provide the basis for the PFOA and PFOS MCLGs, they will likely have large practical impacts and will receive extensive attention and scrutiny. Therefore, the Panel stresses that a strong and transparent rationale for the conclusions about strength of evidence for health outcomes, as well as other components of the assessment, should be provided.

The draft MCLG documents include levels of confidence (i.e., strength of evidence conclusions) in the evidence synthesis sections while the 2016 HESDs did not. The inclusion of such strength of evidence conclusions can clarify the rationale for the selection of health endpoints/systems for consideration for POD development. However, a consistent process for determination of the strength of evidence designations is not clearly described or followed. The sections on Human Evidence and Animal Evidence for the specific health effects/outcomes are often difficult to follow due to the highly summarized presentation of the information and the inconsistent presentation of information and conclusions for different health outcomes (discussed in detail below).

EPA stated that consistent language and a structured approach were not used to describe the level of evidence for different health effects because the health endpoints sections were written by different scientists and the conclusions are based on professional judgment. Specifically, consistent terminology and a consistent approach was not used for the synthesis and integration of human, animal, and mechanistic evidence to make overall hazard conclusions. As discussed in more detail below, the Panel recommends that a consistent approach be used.

The Panel recognizes that due to time and resource constraints the revisions suggested below may not be feasible for all endpoints included in the drafts. The draft MCLG documents, as well

as evaluations by others, indicate that the human evidence for noncancer effects is strongest for effects on the liver, immune system, serum lipids, and fetal growth. The effects with the strongest evidence from animal studies could also be identified; these would likely include hepatic, immune system, and developmental effects, and possibly others. The Panel recommends that EPA focus on revisions to the human, animal, and mechanistic evidence sections and evidence integration sections for those endpoints that have the strongest evidence, rather than for all endpoints. That being said, the Panel recommends that additional endpoints also be evaluated systematically, possibly over a longer timeframe. Conclusions about weight of evidence and PODs for additional endpoints may be important to inform assessments of PFAS mixtures, as described in the draft Mixtures Framework document.

Recommendations

In the short-term and in consideration of the Agency's time constraints, the Panel recommends that EPA initially focus on those health outcomes that have been concluded to have the strongest evidence. Additional health outcomes should also be evaluated using the recommendations below, over a longer timeframe if necessary.

Need for a consistent approach and terminology

Approach for evidence synthesis and integration

The Panel noted that the hazard identification process in the draft MCLG documents is not fully transparent or consistent, either in the evaluation of different evidence "streams" (human, animal, mechanistic), or in their integration. Consistency in the evaluation of human and animal studies, as well as mechanistic data, is important in increasing confidence in the validity of the hazard determinations derived from each of these evidence streams. Several of the public comments also included similar points. For instance, although study quality/risk of bias is discussed, the "synthesis" within an evidence stream does not appear to be structured. Examples of how to structure "synthesis" include the Hill criteria (see IRIS 2020 Handbook and NASEM review of the IRIS Handbook), GRADE, NTP/OHAT, and the Navigation Guide. Similarly, there is no framework for integration of human, animal, and mechanistic data, which is also addressed in several of these examples. These frameworks should include templates for tabular summaries of evidence both "within" and "across" evidence streams that could be adapted to provide more transparency.

The Panel suggests that a format or template be developed so that the information can be presented consistently for each endpoint. For example, an existing framework could be used to make a synthesis conclusion for each evidence stream (human, animal, mechanistic), and an overall hazard conclusion based on integration across the evidence streams. This would provide additional transparency as to the hazard conclusions, as well as clarify the reasons why a particular health endpoint is chosen for candidate RfD derivations. Relevant to this point, Section 2.6 in the Systematic Review section states that modified Hill criteria were discussed for each health outcome, but it appears that this was not actually done in the Hazard Identification section. Inclusion of such a discussion would help to address the issues mentioned above.

Summary tables that present basic information for each study, such as study population (e.g., general population, children, pregnant women, occupational, etc.) for epidemiology studies and lab animal species for toxicology studies, exposure range or dose levels, and overall results for the endpoints evaluated (e.g., significant increase, significant decrease, or no effect) would be helpful to understanding the overall human and animal evidence.

Terms used for strength of evidence designations

Terms such as "suggestive evidence," "moderate evidence," and other seemingly interchangeable terms are used in sections on different health outcomes. These terms are not defined in the draft MCLG documents, and it is unclear whether there is an intended difference among these seemingly similar terms. Additionally, no strength of evidence term is used in some cases, such as, "...the evidence indicates an association between increased serum levels of PFOS and decreased antibody production following routine vaccinations in children," and "EPA concluded the impaired IgM response reported in [animal studies] supported the human results and this endpoint was considered for POD derivation [for PFOA]."

To address these issues, the Panel recommends that consistent descriptors be defined and used for human, animal, and overall strength of evidence conclusions for each endpoint. While recognizing that approaches from the draft 2020 ORD IRIS staff handbook were not used for strength of evidence determinations in the draft MCLG documents, the table of "Evidence integration judgments for characterizing potential human health hazards in the evidence integration narrative" (Table 11-5) in the draft 2020 ORD handbook provides the following descriptors: "evidence demonstrates", "evidence indicates [likely]", "evidence suggests but is not sufficient to infer", "evidence inadequate", and "strong evidence supports no effect." Criteria and examples for applying these descriptors are also provided. Such an approach would be consistent with the use of consistent descriptors and consistent criteria for selection of descriptors for the domains and overall confidence conclusions in the individual study evaluations earlier in the overall process. It would also be consistent with the use of descriptors and criteria for selection of the descriptors for weight of evidence for carcinogenic potential in the U.S. EPA (2005) Guidelines for Carcinogen Risk Assessment.

Consistency in terminology for conclusions about strength of evidence

The Panel also noted inconsistencies in the terminology used to describe the strength of evidence in the health outcomes sections of the draft MCLG documents and how they are applied. For example, in some cases it is stated that there is a certain level of evidence (e.g., suggestive) for "associations of PFOA [or PFOS] with [the effect]", while in other cases, it is stated that there is a certain level of evidence (e.g., suggestive) that PFOA or PFOS "impacts [the effect]" or a certain level of evidence "for [the effect]" (or similar language). It is not clear whether these different terms are intended to distinguish between the level of evidence for an association versus the level of evidence supporting a plausible relationship between PFOA or PFOS and the effect. The Panel recommends that the intended meaning of the strength of evidence conclusions for each health outcome be clarified and that consistent terminology be used for describing these conclusions.

As mentioned above, Section 2.6 in the Systematic Review section states, "*a summary discussion that addresses considerations regarding causation as adapted from Hill (1965)*" is provided for each health outcome. However, such a discussion does not appear to be included for many or most of the health outcomes in the Noncancer Hazard Identification section. The Panel suggests including such a summary discussion of considerations for causation adapted from Hill (1965) to address the issue mentioned above.

Focus on key health outcomes

As above, the Panel recognizes that time limitations may prevent incorporation of the suggestions discussed above for all health outcomes. A possible approach to address this issue would be to focus on those endpoints determined to have the strongest evidence, rather than for all endpoints. That being said, the Panel recommends that additional endpoints should also be evaluated systematically, possibly over a longer timeframe. Conclusions about weight of evidence and PODs for additional endpoints may be important to inform assessments of PFAS mixtures, as described in the draft EPA Mixtures Framework document.

Considering human studies from 2016 HESD

(Note: Some of the comments on this issue are also relevant to the response to the Systematic Review charge question above.)

The Panel concluded that the rationale for not considering human studies which were included in the 2016 HESDs is not clear or supportable. There is no reason to believe that the earlier studies are less relevant or of lesser quality than the newer studies. Consideration of all human studies is especially important because conclusions about the level of evidence for human health effects, which are generally observational rather than experimental, are based on the overall weight of evidence from all relevant data.

The 2016 HESD for PFOA concluded that there was substantial evidence for human effects of PFOA, as follows: "*Human epidemiology data report associations between PFOA exposure and high cholesterol, increased liver enzymes, decreased vaccination response, thyroid disorders, pregnancy-induced hypertension and preeclampsia, and cancer (testicular and kidney)*" and that "...human data identified significant relationships between serum levels and specific indicators of adverse health effects..." The reason that human studies were not considered as the basis for PODs and RfDs in the 2016 HESDs was not related to the quality of the studies themselves or to a lack of overall evidence for human effects, but was rather due to the absence of a toxicokinetic model to relate internal dose (serum levels) to external exposure (administered dose). The 2016 PFOA HESD states that human data were not used as the basis for PODs and RfDs because of "*lack [of] the exposure information for dose-response modeling.*" This is no longer an issue because EPA has now accepted a model (Verner *et al.*, 2016) that can relate serum PFOA and PFOS levels to external dose during developmental lifestages and throughout life, and it has concluded that the human data for PFOA and PFOS can be used in dose-response modeling.

The evaluation of the association of PFOA and high cholesterol in occupationally exposed workers in the draft PFOA document illustrates the need to consider all epidemiology studies,

including those considered in the 2016 HESD, in making strength of evidence conclusions. As stated in Section 3.3.5.1.2.1, the 2016 HESD concluded that there was "*relatively consistent and robust*" evidence of an association of PFOA and increased serum cholesterol in the occupational worker studies. However, the draft PFOA document reviewed only the three newer occupational worker studies not included in the 2016 HESD, all of which were rated as "low confidence." The draft PFOA document states that these newer studies "*suggest no association between PFOA and TC in workers*" and that "*differences in findings from occupational studies between the 2016 [HESD] and this review may be attributable to the limitations of occupational studies in this review.*" There does not appear to be a supportable rationale for making a conclusion based on only three low confidence studies when other potentially stronger studies are also available. As such, the Panel recommends that EPA also consider both the older and newer worker studies to strengthen the overall of weight of evidence conclusion.

Another important aspect of this issue is that specific human epidemiology studies included in the 2016 HESDs may be preferable to the more recent studies for POD development for some health endpoints. As an illustration of this point, the PFOA and PFOS RfD are based on a dataset from Grandjean *et al.* (2012), a study that was included in the 2016 HESDs. Specifically, the RfDs are based on serum PFOA or PFOS levels at age 5 and tetanus or diphtheria vaccine antibody concentrations at age 7 in Cohort 3 (born 1997-2000) from Grandjean *et al.* (2012). Subsequent studies from the same research group (Mogensen *et al.*, 2015; Grandjean *et al.*, 2017) re-published this dataset and/or used this dataset in some of their analyses, but the BMDL (from Budtz-Jorgensen and Grandjean, 2018) that was used as the POD for the final RfD comes only from the dataset originally presented in Grandjean *et al.* (2012). If the subsequent papers that included the dataset from Grandjean *et al.* (2012) in additional analyses had not been published, the Grandjean *et al.* (2012) study and dataset used as the basis for the RfDs would not have been considered for POD development. This example clearly demonstrates why the older human studies included in the 2016 HESD should be considered.

Additional details relevant to this point are: Grandjean *et al.* (2012) is not included in the table summarizing epidemiology studies (Table C-7) in the draft MCLG documents, and there is no systematic review evaluation of Grandjean *et al.* (2012) in HAWC. If the dataset from Grandjean *et al.* (2012) remains the basis for the PFOA and PFOS RfDs, the Panel suggests that a HAWC systematic review of this study be performed and included in the final document.

Consideration of mechanistic information

Evaluation of the mode(s) and/or mechanism(s) of action for toxicity is normally an important part of toxicity assessments such as these draft MCLG documents. However, there is frequently little or no discussion of mechanistic information in the draft MCLG documents, with only summary tables of the number of studies with each type of mechanistic information that were identified, a reference to the section in the 2016 HESD on this topic, and a statement that an updated evaluation will be completed after the SAB review. Mechanistic/mode of action data can help to inform conclusions about the human relevance of effects observed in animal species and the plausibility of effects observed in humans, among other areas of potential uncertainty. The Panel recommends that an evaluation of mechanistic/mode of action data be included for those effects considered as the potential basis for the RfDs, or, at a minimum, for the effect(s) selected

as the basis for the final RfD.

Recommendations

The Panel recommends that a consistent framework and descriptors be used for evidence synthesis and integration for each health outcome. A format or template should be developed so that the information is presented consistently for each endpoint, and consistent descriptors should be defined and used for human, animal, and overall evidence.

The Panel recommends that studies, particularly human studies, that were included in the 2016 HESDs be considered in the same manner as the more recent studies. There is no reason to believe that the earlier studies are less relevant or of lesser quality than the newer studies. Consideration of all human studies is especially important because conclusions about the human health effects, which are generally observational rather than experimental, are based on the overall weight of evidence and should include all relevant data.

The Panel recommends that an evaluation of mechanistic/mode of action data be included for those effects considered as the potential basis for the RfDs, or, at a minimum, for the effect(s) selected as the basis for the final RfD(s).

Selection of endpoints and studies for POD development

Selection of endpoints for POD derivation

In general, the rationale and criteria for the selection of endpoints to derive a POD are not always clearly presented. PODs can potentially be used as the basis for RfDs; therefore, it is important to clearly demonstrate that the endpoints selected for POD derivation are well established, sensitive, adverse or precursor to adverse effects, and that endpoints from animal studies are relevant to humans.

Importantly, the Panel noted that the draft MCLG documents do not clearly distinguish the process of hazard identification from the process of dose-response assessment. For instance, the Evidence integration sections discuss POD derivations as part of Evidence synthesis. Separating the steps of hazard identification and identifying PODs would provide greater transparency. In particular, hazard identification should not depend on whether the data can provide PODs, but rather only health effects with a certain level of confidence in the “hazard” are considered for dose-response. This separation of hazard and dose-response is important because studies that provide strong evidence for hazard do not necessarily need to be amenable to POD derivations (e.g., epidemiologic studies with semi-quantitative exposure metrics), and conversely, the availability of studies amenable to POD derivations is not sufficient to provide strong evidence of hazard. The first step is determining that there is sufficient strength of evidence for “hazard” (independent of dose) to support POD derivation. Then, when selecting studies for POD derivation, determine both whether studies are amenable to POD derivation (especially BMD analysis), as well as study confidence/quality indicators.

Furthermore, the Panel noted that the information about POD derivation for effects described as having "suggestive" evidence in different sections of the draft MCLG documents is internally inconsistent. The documents state (PFOA - p. 317; PFOS – p. 290) that human studies showing an "association" for an effect were used for PODs, but it then goes on to indicate that human health effects with "suggestive" evidence were not used for PODs, as follows: *“Well-conducted ... human studies were prioritized for POD derivation and compared to PODs derived from animal data when possible when the human data provided an association between PFOA and an adverse effect. Such human studies were available for immunotoxicity, developmental, serum lipid, and hepatic effects. For other health effects where the epidemiological data were suggestive of adverse effects, dose response data from the animal studies were prioritized.”* Although the text quoted above indicates that human health effects with suggestive evidence were not used for PODs, the human health effects that were used for PODs (immune, developmental, serum lipids [for PFOA], hepatic) were described as having "suggestive" or "moderate" evidence; a conclusion of "strong" evidence was provided only for PFOS exposure and increased serum lipids. While recognizing that EPA did not use approaches from the draft 2020 ORD staff handbook for strength of evidence determinations in the draft MCLG documents, it should be noted that the draft 2020 handbook does not recommend that PODs for health endpoints with “suggestive” evidence be used as the basis of toxicity values (i.e., Reference Doses). Because of the uncertainty associated with such PODs, the draft 2020 handbook recommends that they be used only for range finding and prioritization.

To address this issue, the Panel recommends that EPA clarify the internal inconsistencies mentioned above. Additionally, the Panel suggests that EPA consider reevaluating its strength of evidence conclusions for some human endpoints, including (but not necessarily limited to) decreased immune response, increased liver enzymes, increased serum lipids (for PFOA), and decreased fetal growth to determine if they are better described as having “likely” or "strong" evidence rather than “suggestive” or "moderate" evidence. A conclusion of "likely" or "strong" evidence would provide additional support for development of a POD to be used as the basis for the RfD.

Based on the draft 2020 handbook criteria for “likely” evidence,¹ which can be selected in certain circumstances when animal and/or human evidence are only “moderate,” and the assumption that “strong evidence” mentioned in the charge question is equivalent to “evidence demonstrates” or “likely evidence” in the draft 2020 handbook, it appears that there may be sufficient evidence to classify additional endpoints as "strong" or “likely.” This is particularly

¹ For “evidence indicates (likely)”, there should be “an evidence base that indicates that [PFOA] exposure likely causes [the health effect] in humans, although there may be outstanding questions or limitations that remain...,” and “this conclusion level is used if there is robust animal evidence supporting an effect and slight-to-indeterminate human evidence, or with moderate human evidence when strong mechanistic evidence is lacking. This conclusion level could also be used with moderate human evidence supporting an effect and slight or indeterminate animal evidence, or with moderate animal evidence supporting an effect and slight or indeterminate human evidence. In these scenarios, any uncertainties in the moderate evidence are not sufficient to substantially reduce confidence in the reliability of the evidence, or mechanistic evidence in the slight or indeterminate evidence base (e.g., precursors) exists to increase confidence in the reliability of the moderate evidence.”

true if the studies included in the 2016 HESD and more recent studies published after the ending date of the literature search for the draft MCLG documents are considered. As mentioned above, consideration of the earlier studies from the 2016 HESD is necessary for determining the overall weight of evidence for each health outcome.

Using immunosuppression as just one example, the Evidence Integration section (3.3.4.4) for immune effects of PFOA states: "*The evidence of an association between PFOA exposure and immunosuppressive effects in human studies is moderate based on largely consistent decreases in antibody response following vaccination ... in two medium-confidence, overlapping birth cohorts.*" Consideration of studies that evaluated vaccine response in populations in other locations, including older studies from the 2016 HESD and recent studies not included in the current draft MCLG documents (additional studies have been suggested by SAB Panel members), could potentially support the conclusion that the evidence for decreased antibody response to vaccines is stronger than "moderate." Associations of PFOA and PFOS exposure with infectious disease are also relevant to strength of evidence for immunosuppression; this topic is discussed in detail below.

Selection of specific studies for POD development

The Panel recommends that a clearer explanation be provided throughout the MCLG documents as to why specific studies were selected for POD development when there are a number of possible choices. As one example, the BMDL presented in Dong *et al.* (2019) was used by EPA as the POD for increased serum cholesterol. However, no information is provided in the PFOA document as to why Dong *et al.* (2019) was selected for dose-response for serum lipids from the many studies that are available. Furthermore, the BMD modeling section (Appendix B) states that few details are provided by the authors about several aspects of the BMD modeling presented in the Dong *et al.* (2019) publication; this lack of information does not appear to support use of this BMDL as a POD.

It is possible, although not stated in the draft MCLG documents, that Dong *et al.* (2019) was selected because studies included in the 2016 HESD were not considered. Again, the Panel strongly recommends that older studies that were included in the 2016 HESD be considered for POD development, and the Panel notes that some of these older studies were considered in the EPA's draft "Analysis of CVD Risk Reduction as a Result of Reduced PFOA and PFOS Exposure in Drinking Water." As discussed in the response to the Systematic Review charge question above, there is no reason to assume that the more recent studies are preferable to the older studies, as illustrated by the choice of a dataset from Grandjean *et al.* (2012) as the basis for the final RfDs for both PFOA and PFOS in the draft MCLG documents.

Additional comments about the studies and endpoints used for PODs

The list of studies and endpoints considered for POD development in Table 15 is not totally consistent with the information in the text and the dose-response results in Appendix B. For example, some of the studies/endpoints (corpora lutea, body weight) stated in the text of the PFOA document to have been selected for dose-response/POD development were not modeled, while other endpoints (kidney weight) stated not to be selected for dose-response/POD

development were modeled. Also, the Panel noted that the draft PFOA document states that prenatal loss was modeled from Lau *et al.* (2006), but the data from Wolf *et al.* (2007), not Lau *et al.* (2006), were actually modeled.

There is also inconsistency in the evidence integration sections for different health outcomes regarding whether any human and/or animal studies were selected for POD development and, if so, which ones. For example, the human evidence integration discussions for developmental effects do not state whether any of the human studies were selected for POD development, although several birthweight studies were later selected (Table 15) for both PFOA and PFOS. As another example, the evidence integration sections for immune effects in both MCLG documents do not state that a human study of vaccine response (which eventually was used as the basis for the final RfD) was selected for POD development. In contrast, the evidence integration sections for some other endpoints specifically mention whether or not any studies were selected for POD development, and, if so, which studies were selected.

Finally, while it is stated (in the text quoted above - PFOA - p. 317, first paragraph; PFOS – p. 290, final paragraph) that PODs were developed from human data for hepatic effects, human data for hepatic effects was not used for POD development (as stated for PFOA on p.148, for PFOS on p. 133, and in the charge question below).

Recommendations

The Panel recommends that the process of hazard identification be separated from the process of dose-response assessment. A conclusion about evidence of hazard should not depend on whether or not the data can provide PODs. Instead, sufficient evidence for hazard is needed before dose-response assessment for a health outcome can be considered.

The Panel recommends that the rationale and criteria for selection of endpoints and specific studies for POD development be more clearly presented. It is important to clearly demonstrate that the endpoints selected for POD development are well established, sensitive, adverse or precursor to adverse, and that endpoints from animal studies are relevant to humans. Internal inconsistencies in the criteria used for selection of endpoints for POD development should be addressed. It is also important to explain why a specific study of a health endpoint was selected when there are several possible choices.

Strength of evidence designations for specific health outcomes

This section of the response provides comments on discussions and conclusions on specific noncancer health outcomes in the Hazard Identification sections of the draft MCLG documents. The comments above on general issues with the determination of strength of evidence are also relevant to this part of the response.

Evidence for effects on the immune system, liver, fetal growth, and serum lipids

The Panel agrees with EPA that the most consistent epidemiological associations with PFOA and PFOS exposure are for decreased immune system response, decreased fetal growth (i.e.,

1 decreased birthweight), increased serum lipids, and increased liver enzymes, particularly
2 Alanine Aminotransferase (ALT). Multiple studies for each of these four effects are generally
3 consistent in different populations and settings, and the total body of evidence indicates that
4 these effects are present. While there is no single definitive study for any of these endpoints
5 (which is not a realistic goal), multiple studies of adequate quality pointing in the same direction
6 justifies the conclusion that PFOA and PFOS are associated with these health endpoints.
7 Additional evidence from animal studies further supports the conclusion that PFOA and PFOS
8 cause these effects. The available data is more limited and/or the evidence is less consistent for
9 associations of PFOA and PFOS with other health endpoints including effects on the thyroid,
10 ulcerative colitis, neurodevelopmental effects, and others.

11 For the four consistent endpoints, most studies report relatively small changes in clinical
12 biomarkers. While most of these studies did not evaluate the number of subjects with a
13 clinically abnormal value for biomarkers, one or more studies, for each of the four effects,
14 reported an association of PFOA and/or PFOS with increased risk of a clinically abnormal
15 value. Examples of studies that reported an increased risk of clinically abnormal values are:
16 tetanus or diphtheria antibodies levels below a clinically protective level - Grandjean *et al.*
17 (2012) and possibly others; clinically defined low birth weight or small for gestational age -
18 multiple studies reviewed in the draft MCLG documents; clinically defined high cholesterol -
19 Steenland *et al.* (2009) and possibly others; clinically defined elevated ALT - Gallo *et al.*
20 (2012) and Darrow *et al.* (2016). In studies where ALT was not specifically evaluated, an
21 increase in the number of subjects with a clinically abnormal value is also expected from the
22 overall change (shift in the distribution curve) in the abnormal direction. While the clinical
23 relevance of exposure to PFOA or PFAS cannot be predicted on an individual basis, the
24 increased number of individuals within a population with clinically defined abnormal values is
25 of public health concern.

26 An increase in the clinical diseases related to the biomarkers mentioned above has not been
27 consistently demonstrated for several of these endpoints. As discussed in the response to the next
28 charge question, the limited available information does not demonstrate an increase in liver
29 disease although data on hepatic effects of PFAS in animals and humans indicate that additional
30 research on PFAS and liver disease is needed. Similarly, as discussed in the draft PFOA
31 document, the limited available evidence does not demonstrate an association with
32 cardiovascular disease and more research is needed. As mentioned above, multiple studies
33 support an association of PFOA and PFOS with clinically defined low birth weight/small for
34 gestational age. However, the Panel is not aware of evidence for associations of PFOA and
35 PFOS with adverse consequences such as developmental delays in low birth weight/small for
36 gestational age infants.

37 Regarding associations of PFOA and PFOS and infectious disease, a recent review that focused
38 on PFOS (Pachkowski *et al.*, 2019) concluded that studies available through 2018 "provide
39 evidence for an association between general population levels of PFOS exposure and infectious
40 disease, a clinical meaningful measure of health risk," and another review that focused on
41 PFOA (Steenland *et al.*, 2020) concluded, "evidence that PFOA increases risk of human
42 infectious disease is inconsistent." The Human Evidence subsections of the Immune sections
43 outline the findings from studies of PFOA and PFOS and infectious disease. While results of

these studies are not consistent, many of them reported associations with infectious disease. However, the Evidence Integration subsections of the Immune sections neither provide conclusions on associations with infectious disease nor mention infectious disease; the reason for this omission is unclear, especially since immunosuppression is the critical effect for the RfD. Additionally, several recent studies, not cited in the draft MCLG documents (e.g., Timmermann *et al.*, 2020; Dalsager *et al.* 2021; Bulka *et al.*, 2021), report associations for both PFOA and PFOS with increased risk of infectious disease. The Panel recommends that weight of evidence conclusions be developed for infectious disease in the revised documents.

Clarifications in Hazard Identification information about serum cholesterol

The Panel concluded that it is especially important to provide a clear and thorough discussion of the strength of evidence for the association of PFOA and PFOS and increased serum cholesterol since this effect is a major part of the basis for the separate evaluation of cardiovascular disease risk. The draft MCLG documents state that PFOA and PFOS caused decreased serum lipids in some animal studies while lipids are increased in human studies. They further state that this interspecies difference may be due to the "*difference in serum lipid composition between humans and commonly used rodent models*" and that "*food consumption may confound these results, as diet is a major source of lipids, yet studies do not consistently report a fasting period before serum collection.*"

Other potentially important explanations for this human versus animal difference in the effect of PFOA and PFOS on serum lipids should be discussed in the draft MCLG documents. These include much lower human exposure levels compared to the doses used in animal studies (see discussion below), differences in the fat content of human diets versus rodent lab diet, and differences in the activity of PPAR- α in humans and laboratory animals. Studies that investigated these issues were reviewed in DWQI (2017) including Tan *et al.* (2013) and Rebholz *et al.* (2016), as well as newer studies such as Schlezinger *et al.* (2020).

In general, human and rodent data suggest that the effects of PFOA and other PFAS on lipid formation and storage results from the balance of different effects which may act in opposite directions (Das *et al.*, 2017). The decrease in serum lipids at the higher doses used in animal studies is believed to be due to activation of PPAR- α (DWQI, 2017). PPAR- α is also active in humans, as demonstrated by the use of PPAR- α activating drugs to decrease high cholesterol in humans. However, PFOA, PFOS, and other PFAS do not activate PPAR- α in humans at lower environmentally relevant doses, and the increased serum lipids associated with PFOA, PFOS, and other PFAS may result from activation of other receptors and/or biological pathways involved with lipid metabolism that act in the opposite direction.

Convertino *et al.* (2018) is a study of advanced cancer patients who were given extraordinarily high doses of PFOA. The draft PFOA document acknowledges concerns and limitations of this study. It states that "*participants dosed with extremely high levels of ammonium perfluorooctanoate (APFO), a PFOA precursor, in an open-label, nonrandomized, phase 1 trial, were found to have reduced levels of total cholesterol with increasing plasma PFOA concentrations.*" It further states that Convertino *et al.* (2018) "*differed from the other studies in several ways. First, all participants were solid-tumor cancer patients who failed standard*

1 *therapy. Second, participants ingested APFO rather than being exposed to PFOA. Third,*
2 *participants' plasma PFOA concentrations were several orders of magnitude higher than those*
3 *reported in the general population," and that "it is unclear if these factors explained the inverse*
4 *association between PFOA and total cholesterol."* This study was rated as "low confidence" by
5 EPA.

6 While Convertino *et al.* (2018) may arguably fulfill the PECO criteria for health effects studies
7 of PFOA in humans shown in Table 2, the Panel concluded that this study does not appear to be
8 appropriate for consideration in hazard identification of PFOA. Some of these concerns were
9 described in the NJDEP (2020) document which states, Convertino *et al.* (2018) "*is not useful in*
10 *the evaluation of potential health effects of chronic drinking water exposure to PFOA in the*
11 *general population," and that "limitations of this study include small sample size, very short*
12 *length, limited power of study, and potential altered metabolic state of study group consisting of*
13 *late-stage cancer patients. Observations in these patients cannot be considered relevant to*
14 *healthy individuals because their nutritional and physiological status was likely affected by their*
15 *severe illness."*

16 The draft PFOA document acknowledges some of these problematic issues with Convertino *et al.*
17 *et al.* (2018). If consideration of this study remains in the PFOA document, the Panel suggests that
18 the additional information below about potential PFOA-related toxicity in the study, the dose-
19 response for increased cholesterol, and the lack of relevance of the possible increase in serum
20 cholesterol to environmental exposures be included, as follows.

21 An earlier abstract about this study (Macpherson *et al.*, 2010) stated that one of the patients
22 dosed experienced drug related toxicity (DLT) consisting of "grade 5 renal failure and
23 transaminitis" (indicative of liver damage), and that these effects were noted as "possibly drug
24 related." This indicates the potential for PFOA to cause renal and hepatic toxicity in humans,
25 and that it is unclear why the observation of "possibly drug related" kidney and liver toxicity
26 reported by Macpherson *et al.* (2010) was not mentioned by Convertino *et al.* (2018).

27 Also, as mentioned above, the plasma PFOA levels in the subjects in this study were
28 extraordinarily high. The plasma PFOA levels in the 10 exposure categories shown in Figure 4 of
29 Convertino *et al.* (2018) ranged from ~4000 ng/ml to ~630,000 ng/ml. Cholesterol was decreased
30 only in the three highest exposure categories (approximately 262,000 ng/ml or higher plasma
31 PFOA), but not in the seven lower exposure categories that also had extremely high plasma
32 PFOA levels of up to approximately 200,000 ng/ml. The plasma PFOA levels at which
33 cholesterol was decreased are many orders of magnitude above those found in the general
34 population or in communities with contaminated drinking water. They are higher than the highest
35 serum or plasma PFOA levels in occupationally exposed workers in data summarized in Table 5-
36 27 of ATSDR (2021), and they are similar to the serum PFOA levels at which cholesterol is
37 decreased in animal studies, presumably through activation of PPAR- α . The observation of
38 decreased cholesterol at these extremely high plasma concentrations is consistent with the effects
39 of PPAR- α activating drugs that reduce serum cholesterol in humans. In contrast, the increased
40 cholesterol associated with PFOA in the general population and in individuals exposed through
41 contaminated drinking water likely occurs through a different mechanism that is operational at
42 much lower PFOA concentrations.

Consistency in consideration of medications that may affect health outcomes

The Panel also suggests that EPA be consistent in consideration of the use of medications that may affect health outcomes. Specifically, the use of blood pressure medications was not considered in evaluation of blood pressure studies, while studies of serum lipids were rated as deficient if they did not consider lipid lowering medications.

Additional endpoints that should be considered for POD derivation

The list of health effects/endpoints considered appears to be generally complete and appropriate. The Panel noted that the health effects considered for POD derivation have been expanded since the 2016 HESDs from less than 10 studies, all of which were animal toxicology studies, to more than 20 studies, that include both animal toxicology and human epidemiology studies, in the current draft.

Recommendations

The Panel recommends that EPA consider reevaluating its strength of evidence conclusions for some human endpoints, including (but not necessarily limited to) decreased immune response, increased liver enzymes, increased serum lipids (for PFOA), and decreased fetal growth to determine if they are better described as having “likely” or “strong” evidence rather than “suggestive” or “moderate” evidence of an association with exposure to PFOA/PFOS. Such a reevaluation should consider studies included in the 2016 HESD and more recent studies published after the end date of the literature search for the current draft.

The Panel specifically recommends that issues related to the strength of evidence for PFOA and PFOS exposure and increased serum cholesterol be discussed clearly and thoroughly, including but not limited to the specific issues discussed in this response. This is particularly important because this effect is a major part of the basis for the separate evaluation of cardiovascular disease risk.

Charge Question #2B. Elevation of ALT

Elevation of liver serum biomarkers in humans is frequently used as an indication of liver injury, although it has not been shown to be as specific as functional tests, such as histology findings and liver disease (Boone, 2005, HERO ID: 782862). However, greater than 2-fold increases in alanine aminotransferase (ALT) activity, the most sensitive test of hepatocellular injury in humans, above the upper limit of normal are considered indicative of hepatocellular injury. EPA concluded that the available data in adults show a consistent positive association between PFOA and/or PFOS exposure and increased serum ALT levels in the epidemiological literature. However, this response was not selected for dose response modeling because 1) the magnitude of the effect was not large compared to control levels; and 2) concerns about the clinical relevance of the findings and non-specificity of the biomarkers relationship to adverse liver injury and disease.

1 *Does the SAB panel agree with EPA's rationale for not considering the ALT endpoint*
2 *reported in the epidemiological studies for the derivation of a POD for the liver health*
3 *effects? Please provide your justification and if you suggest that EPA consider this endpoint*
4 *for POD derivation, please provide your recommendations for a modeling approach.*

- 5 *i. Are you aware of additional studies that support the ALT levels as markers of adverse liver*
6 *effects? Please provide citations.*
7 *ii. Are there other adverse liver endpoints identified in the epidemiological literature that*
8 *need to be considered?*

9 **Consideration of ALT as an endpoint for PFOA and/or PFOS**

10 The Panel does not agree with EPA's rationale for not considering the ALT endpoint reported in
11 the epidemiological studies for the derivation of a POD for the liver health effects for the
12 following reasons:

13 The draft EPA documents do not provide a compelling reason for considering the changes in
14 ALT to be substantially less supported than other endpoints such as changes in birth weight,
15 serum lipids, or antibody responses. None of these are clinical disease endpoints, but all of them
16 are indicative of an increased risk for adverse health effects, and all have a similar level of
17 evidence to the effects due to changes in ALT. The elimination of ALT as an endpoint for POD
18 derivation therefore does not appear to be totally consistent with the rationale for developing
19 PODs for some other human health effects or with EPA risk assessment guidance. As stated in
20 the charge question, increased ALT is indicative of liver damage, and U.S. EPA (2002)
21 guidelines for RfD development state that a RfD should be based on an adverse effect or a
22 precursor to an adverse effect. The Panel noted that, although the magnitude of PFOA's effect on
23 ALT may not be large, the same may also be true for the magnitude of the PFOA's effects on
24 other human health endpoints such as, increased cholesterol and decreased birth weight. As such,
25 if a POD is not developed for the ALT endpoint, an explanation should be provided as to why
26 the magnitude of the effect was not sufficient for ALT but was sufficient for other effects of
27 similar magnitude.

28 Numerous observations in human epidemiological studies that associate serum PFOA and PFOS
29 concentrations with serum ALT, and multiple rat and mouse studies that also demonstrate PFOA
30 and PFOS administration can raise serum ALT, consistent with the epidemiological studies. This
31 observation has been made by numerous research groups in more than one animal species, such
32 that ALT represents a reproducible and rigorous endpoint that is predictive of adverse health
33 effects. In fact, California EPA (2021) selected increased risk of clinically elevated serum ALT
34 as the basis for its draft RfD for PFOA. In their evaluation, California EPA (2021) considered the
35 issues related to use of elevated ALT in humans as a critical effect that are discussed in the
36 charge question above. The Panel suggests that EPA consider the California EPA (2021)
37 rationale for its decision to use elevated ALT as the critical effect for RfD development. If EPA
38 decides to develop a POD for elevated ALT, the Agency should also consider the modeling
39 approach used by California EPA for this effect.

Regarding clinically defined elevated ALT, the draft EPA PFOA document discusses Darrow *et al.* (2016), which reported an association between modeled PFOA exposure and increased risk of clinically elevated ALT in a population with elevated PFOA exposure from contaminated drinking water. Gallo *et al.* (2012), which was not included in the draft PFOA document but was reviewed in the 2016 HESD, also found an association between measured serum PFOA levels and increased risk of clinically elevated ALT in the same study group.

Furthermore, as noted in a recent review of epidemiological evidence for health effects of PFOA by Steenland *et al.* (2020): "There is also evidence that effects on ALT are more pronounced among obese subjects, who are at higher risk of nonalcoholic fatty liver disease (Lin *et al.* 2010; Jain and Ducatman 2019)."

Also relevant to the potential significance of elevated ALT, Steenland *et al.* (2020) concluded that while "the limited existing evidence does not support a link between PFOA and diagnosed liver disease," there is a lack of "adequately powered epidemiologic studies of liver disease" and PFOA. They stated that "the established liver toxicity of PFOA in experimental animal studies ..., the storage of PFOA in liver tissue in humans, and extensive evidence that PFOA exposure is associated with markers of hepatocyte cell death, warrants additional research on PFAS and liver disease, particularly nonalcoholic fatty liver disease."

Recommendations

Accordingly, the Panel recommends that Gallo *et al.* (2012) and other epidemiological studies of liver enzymes that were included in the 2016 HESD, as well as any new studies identified in the literature review, be considered when evaluating the weight of evidence for epidemiological effects of PFOA and PFOS as well as for POD derivation.

Studies that support the ALT levels as markers of adverse liver effects

There are numerous examples of associations of elevated ALT with disease endpoints, as well as <2-fold increase in serum ALT being associated with pathology-confirmed liver disease, such as non-alcoholic fatty liver disease (NAFLD). Importantly, the AASLD has a position paper (Kim *et al.* 2008) about serum ALT being considered a predictor for overall health and mortality. Thus, the Panel views it as inappropriate to conclude that less than a 2-fold change in serum ALT is innocuous and without risks to human health.

According to the American College of Gastroenterology (ACG), "Multiple studies have demonstrated that the presence of an elevated ALT has been associated with increased liver-related mortality." Kwo *et al.*, 2017 state the following: There is an accumulating set of data demonstrating that AST and ALT elevations correlate with morbidity and mortality. An initial report from Germany noted that those with AST>18 U/l had a 3X increased risk of all-cause mortality. A Korean study found that, compared to men with AST or ALT<20 IU/l, the 30–39 IU/l group had an 8X (AST) or 9.5X (ALT) relative risk (RR) for liver-related death. Similar results were demonstrated from a study comparing the standardized mortality ratios in subjects from Olmsted County where higher ALT levels correlated with higher mortality with the standardized mortality ratio being 0.95 for normal ALT (defined as ULN (i.e., upper limit of

normal) 45 IU/l for men, 29 IU/l for females), 1.32 for 1–2X ULN, and 1.78 for >2X ULN with a similar relationship for AST levels.

Studies have used the data from the National Health and Nutrition Examination Survey (NHANES) databases to assess risk of morbidity and mortality in relationship to abnormal liver tests with one study demonstrating that elevated ALT (ULN defined as 30 U/l for men and 19 U/l for women) was associated with significant increases in liver-related mortality (11.2X) and diabetes-related mortality (3.3X). Another analysis demonstrated that ALT>43 IU/l for men and >30 IU/l for women was related to the presence of coronary heart disease, even when patients with obesity, chronic viral hepatitis, and excessive alcohol use were excluded.

Author /year	Proposed ALT/AST cutoff level	ALT/AST level for increased mortality	Comments
Arndt <i>et al.</i> (27)	AST 18	AST>18	3X increase in all cause mortality
Kim <i>et al.</i> (28)	ALT<20	ALT 30–39	RR of liver mortality 2.9 (2.4–3.5) and 9.5 (7.9–11.5) in men, 3.8 (1.9–7.7) and 6.6 (1.5–25.6) in women
Lee <i>et al.</i> (29)	ALT (ULN 45 IU/l for M, 29 for F)	ALT 45–90 M 29–58 for F	SMR risk 1.32 for 1–2X ULN, and 1.78 for >2X ULN
Ruhl and Everhart (30)	ALT 30 IU/l M, 19 IU/l for F	ALT>30 for M ALT >19 for F	Increased liver related mortality
ALT, alanine aminotransferase; AST, aspartate aminotransferase; F, female; RR, relative risk; M, male; ULN, upper limit of normal.			

The American Association for the Study of Liver Diseases (AASLD) guidance regarding serum ALT levels as an indicator of health and disease article argues that serum ALT is more than a marker for liver injury, and that ALT “may also be a good indicator of overall health, particularly in the context of obesity, the metabolic syndrome, and presence of cardiovascular disease, as many patients affected by these conditions also are at risk of having non-alcoholic fatty liver disease”.

With regard to asymptomatic liver disease, a Scandinavian study, Mathiesen *et al.*, 1999, of 151 consecutive patients with mild to moderate elevations of serum aminotransferase levels for at least 6 months in combination with liver biopsy revealed that non-alcoholic steatohepatitis and hepatic steatosis (noted in 42%), chronic HCV (15%), alcoholic liver disease (8%) and autoimmune hepatitis, primary biliary cirrhosis and alpha1 antitrypsin deficiency was associated with only slight to moderate elevation in serum ALT.

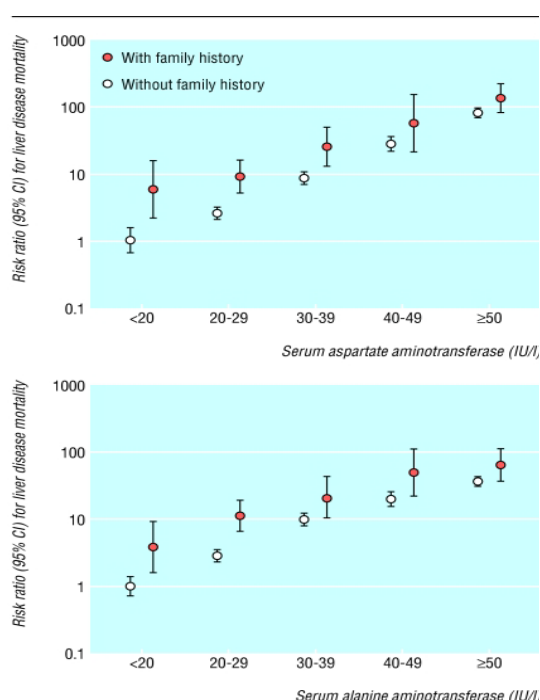
Kim *et al.*, 2004 reported a population-based study that included 142,055 individuals of ages between 35 and 59 years in whom baseline demographic and laboratory data obtained between 1990 and 1992 were available that found a positive association between the aminotransferase concentration and mortality from liver disease in serum ALT values in normal range (35–40

IU/l). The risk ratio for liver disease mortality also correspondingly increased as serum ALT increased.

In Kim *et al.* (2008), of 47,182 county residents in Minnesota who had healthcare encounters in 1995, 6,823 (14.5%) had their ALT measured, of which 5,912 had results within normal limits and 911 (13.4%) abnormal. Abnormal AST was associated with a significantly increased standardized mortality ratio (1.32 for 1-2x ULN and 1.78 for >2x ULN). Ruhl *et al.* (2013) used NHANES 1988–1994 data and the mortality risk of persons in ALT deciles 1, 2, 3, and 10 was compared with that of persons in deciles 4–9 (mortality was relatively flat across these deciles) over an 18-year period (through 2006) among 14,950 viral-hepatitis-negative adults. High ALT was associated with increased mortality risk.

Kim *et al.*, *BMJ*, 328(7446):983, 2004.

Figure 1.



Moreover, there are case examples of normal ALT in the presence of histologic Non-alcoholic fatty liver disease. For example, in a cohort of 458 subjects, Fracanzani *et al.*, 2008, in Italy, non-alcoholic steatohepatitis was diagnosed in 59% of the patients with normal ALT, respectively. Another cross-sectional study, Park *et al.*, 2015, using Korean NHANES data saw an association with the upper limit of the normal range for ALT with liver disease. They defined the 95th percentile levels for ALT in healthy weight, metabolically normal, liver disease-free KNHANES participants as 34 IU/L for men and 25 IU/L for women. The sensitivity for detecting chronic liver disease was significantly improved when the threshold for normal ALT was lowered.

Recommendations

The Panel recommends the use of ALT as endpoint in light of the numerous studies in the literature support an association between slight elevations in ALT are and increased risk of morbidity and/or mortality. Moreover, these studies suggest that patients with even slight elevations in ALT should be monitored for liver disease. The Panel identified the following citations that appear to be relevant to the issues of the clinical relevance of ALT elevations and of the association of elevated ALT with morbidity and mortality:

Ji L, Cai X, Bai Y, Li T. 2021. Application of a Novel Prediction Model for Predicting 2-Year Risk of Non-Alcoholic Fatty Liver Disease in the Non-Obese Population with Normal Blood Lipid Levels: A Large Prospective Cohort Study from China. *Int J Gen Med*.14:2909-2922

Oh TK, Jang ES, Song IA. 2021. Long-term mortality due to infection associated with elevated liver enzymes: a population-based cohort study. *Sci Rep*.11(1):12490.

Chen J, Liu S, Wang C, Zhang C, Cai H, Zhang M, Si L, Zhang S, Xu Y, Zhu J, Yu Y. 2021. Associations of Serum Liver Function Markers With Brain Structure, Function, and Perfusion in Healthy Young Adults. *Front Neurol*.12:606094. Doi: 10.3389/fneur.2021.606094.

Abdalgwad R, Rafey MF, Murphy C, Ioana I, O'Shea PM, Slattery E, Davenport C, O'Keeffe DT, Finucane FM. 2020. Changes in alanine aminotransferase in adults with severe and complicated obesity during a milk-based meal replacement programme. *Nutri Metab (Lond)*. 17:87. Doi: 10.1186/s12986-020-00512-5.

Newton KP, Lavine JE, Wilson L, Behling C, Vos MB, Molleston JP, Rosenthal P, Miloh T, Fishbein MH, Jain AK, Murray KF, Schwimmer JB. 2021. Alanine Aminotransferase and Gamma-Glutamyl Transpeptidase Predict Histologic Improvement in Pediatric Nonalcoholic Steatohepatitis. Nonalcoholic Steatohepatitis Clinical Research Network (NASH CRN). *Hepatology*.73(3):937-951

Lu Y, Wang Q, Yu L, Yin X, Yang H, Xu X, Xia Y, Luo Y, Peng Y, Yu Q, Chen Z, Yu J, Lai M, Wu N, Pan XB, Zheng X.J. 2020. Revision of serum ALT upper limits of normal facilitates assessment of mild liver injury in obese children with non-alcoholic fatty liver disease. *Clin Lab Anal*. 34(7):e23285. Doi: 10.1002/jcla.23285.

Wahlang B, Appana S, Falkner KC, McClain CJ, Brock G, Cave MC. 2020. Insecticide and metal exposures are associated with a surrogate biomarker for non-alcoholic fatty liver disease in the National Health and Nutrition Examination Survey 2003-2004. *Environ Sci Pollut Res Int*. 27(6):6476-6487.

Park JH, Choi J, Jun DW, Han SW, Yeo YH, Nguyen MH.J. 2019. Low Alanine Aminotransferase Cut-Off for Predicting Liver Outcomes; A Nationwide Population-Based Longitudinal Cohort Study. *Clin Med*. 8(9):1445.

- 1 Schmilovitz-Weiss H, Gingold-Belfer R, Grossman A, Issa N, Boltin D, Beloosesky Y, Morag
2 Koren N, Meyerovitch J, Weiss A. 2019. Lowering the upper limit of serum alanine
3 aminotransferase levels may reveal significant liver disease in the elderly. *PloS One*.
4 14(4):e0212737. Doi: 10.1371/journal.pone.0212737.
- 5 Kim HR, Han MA. 2018. Association between Serum Liver Enzymes and Metabolic Syndrome
6 in Korean Adults. *Int J Environ Res Public Health*. 15(8):1658.
- 7 Umehara T. 2018. Nonalcoholic fatty liver disease with elevated alanine aminotransferase levels
8 is negatively associated with bone mineral density: Cross-sectional study in U.S. adults. *PloS*
9 *One*.13(6):e0197900.
- 10 Kim, WR, Flamm, SL, Di Bisceglie, AM, and Henry C. Bodenheimer Jr. 2008. Serum activity of
11 alanine aminotransferase (ALT) as an indicator of health and disease, *Hepatology*,
12 47(4): 1363-1370; <https://doi.org/10.1002/hep.22109>.
- 13 Kim WR, Benson JT, Therneau TM, Burritt MF, Melton LJ. 2008. Serum aminotransferase
14 activity and risk of mortality in a U. S. community population. *Hepatology*; 47. DOI:
15 10.1002/hep.22090.
- 16 Ruhl C.E., Everhart J.E. The Association of Low Serum Alanine Aminotransferase Activity
17 With Mortality in the US Population. *American Journal of Epidemiology*, 12:2013, p1702–1711,
18 178. <https://doi.org/10.1093/aje/kwt209>.

20 **Consideration of other endpoints**

- 21 Members of the Panel also suggested the inclusion of non-alcoholic fatty liver
22 disease/steatosis as an adverse liver endpoint. As noted by Steenland *et al.* (2020), a recent
23 study of adults from a community with elevated exposure to PFOA from contaminated
24 drinking water showed that PFOA "was associated with cytokeratin 18 M30, a marker of
25 hepatocyte apoptosis (Bassler *et al.*, 2019), and a mechanism of disease progression in
26 nonalcoholic fatty liver disease." Bassler *et al.* (2019) provides further evidence that PFOA
27 causes liver cell injury, and the Panel suggests that EPA consider this study, in addition to
28 other relevant references, including:
- 29 Bassler J, Ducatman A, Elliott M, Wen S, Wahlang B, Barnett J, Cave MC. 2019. Environmental
30 Perfluoroalkyl Acid Exposures Are Associated with Liver Disease Characterized by
31 Apoptosis and Altered Serum Adipocytokines. *Environ Pollut*. 247:1055-1063. doi:
32 10.1016/j.envpol.2019.01.064. PMID: 30823334, PMCID: PMC6404528.
- 33 Jin R, McConnell R, Catherine C, Xu S, Walker DI, Stratakis N, Jones DP, Miller GW, Peng C,
34 Conti DV, Vos MB, Chatzi L. 2020. Perfluoroalkyl Substances and Severity of Nonalcoholic
35 Fatty Liver in Children: An Untargeted Metabolomics Approach. *Environ Int*. 134:105220.
36 doi: 10.1016/j.envint.2019.105220. PMID: 31744629, PMCID: PMC6944061.

Wahlang B, Jin J, Beier JI, Hardesty JE, Daly EF, Schnegelberger RD, Falkner KC, Prough RA, Kirpich IA, Cave MC. 2019. Mechanisms of Environmental Contributions to Fatty Liver Disease. *Curr Environ Health Rep.* 6(3):80-94. doi: 10.1007/s40572-019-00232-w. PMID: 31134516, PMCID: PMC6698418. (Table 1 of this manuscript contains additional citations exploring fatty liver disease and associations with PFOA/PFOS).

Charge Question # 3- Cancer Designation

A. PFOA: Based on new cancer studies identified since the 2016 PFOA Health Advisory (HA), EPA concludes that the available cancer data for PFOA indicate a 'likely carcinogen' categorization which is a change from 'suggestive' in the 2016 HA. Does the panel agree with the 'likely' designation based on the new evidence? If yes, is the rationale clearly described? If no, please provide an explanation for arriving at a different conclusion.

Designation of PFOA as a "likely carcinogen"

EPA *Guidelines for Carcinogen Risk Assessment* (2005) provide a somewhat structured approach for assessing the weight of evidence regarding carcinogenic potential of an agent and for designation as: carcinogenic to humans, likely to be carcinogenic to humans, suggestive evidence of carcinogenic potential, inadequate information to assess carcinogenic potential, or not likely to be carcinogenic to humans.

Based on EPA's *Guidelines for Carcinogen Risk Assessment* (2005), supporting data for the "likely" descriptor may include:

- "An agent demonstrating a plausible (but not definitively causal) association between human exposure and cancer, in most cases with some supporting biological, experimental evidence, though not necessarily carcinogenicity data from animal experiments";
- "An agent that has tested positive in animal experiments in more than one species, sex, strain, site, or exposure route, with or without evidence of carcinogenicity in humans";
- "A positive tumor study that raises additional biological concerns beyond that of a statistically significant results, for example, a high degree of malignancy, or an early age at onset";
- "A rare animal tumor response in a single experiment that is assumed to be relevant to humans"; or
- "A positive tumor study that is strengthened by other lines of evidence, for example, either plausible (but not definitively causal) association between human exposure and cancer or evidence that the agent or an important metabolite causes events generally known to be

associated with tumor formation (such as DNA reactivity or effects on cell growth control) likely to be related to the tumor response in this case."

In general, the Panel agrees that: a) the evidence for potential carcinogenicity of PFOA has been strengthened since the 2016 HESD; b) the results of human and animal studies of PFOA are consistent with the examples provided above and support a designation of "likely to be carcinogenic to humans"; and c) the data exceed the descriptors for the three designations lower than "likely to be carcinogenic".

The Panel noted that EPA's designation for PFOA is consistent with the California EPA (2021) conclusion that PFOA and PFOS should be evaluated as carcinogens for the setting of Public Health Goals (analogous to MCLGs) for PFOA and PFOS. It should be noted that the criteria used by California EPA for determination that a chemical is a carcinogen are not identical to the criteria in the U.S. EPA (2005) *Guidelines for Carcinogen Risk Assessment*.

Supporting epidemiologic and experimental evidence for "likely carcinogenic" designation

Since the publication of the 2016 document for PFOA, at least eight epidemiological studies considering links between PFOA exposure and cancer have been published. The evidence from epidemiologic studies is primarily based on the occurrence of kidney and testicular cancer. At least one new general population nested case-control study conducted in the PLCO cohort (Shearer *et al.*, 2021) supports previous positive associations with kidney cancer observed in individuals highly exposed to PFOA (elevated exposure from contaminated drinking water near PFAS producing facility) (Barry *et al.*, 2013, Vieira *et al.*, 2013). EPA *Guidelines for Carcinogen Risk Assessment* state that, "When human data of high quality and adequate statistical power are available, they are generally preferable over animal data and should be given greater weight in hazard characterization and dose-response assessment, although both can be used." The EPA determined that the new epidemiologic studies were all *medium confidence* studies, including the Shearer *et al.* study, and that there is an absence of any *high confidence* epidemiologic studies. The Panel generally agreed that the epidemiologic evidence is consistent with the example of supporting data for the likely descriptor presented above in the Guidelines, that is, of "plausible associations between human exposure and cancer."

PFOA also caused testicular Leydig cell, pancreatic acinar cell, and/or hepatocellular tumors in male Sprague-Dawley rats in three chronic studies (Butenhoff *et al.*, 2012 [also reported by Sibinski, 1987]; Biegel *et al.*, 2001; NTP, 2020), including one new chronic cancer bioassay in rats (NTP, 2020) which supports previous evidence of tumorigenesis at multiple sites.

In NTP (2020), the incidence of both malignant and benign liver tumors was increased, and the incidence of pancreatic tumors was very high in all dosed groups of males. However, the occurrence of malignant tumors and the very high incidence of pancreatic tumors is not mentioned in the draft EPA PFOA document, and it should be added because it adds to the weight of evidence for PFOA's carcinogenic potential. There was also a marginal increase in hepatocellular carcinomas and uterine adenocarcinomas, and non-significant increases in benign and malignant pancreatic acinar cell tumors, in females in NTP (2020). The lower response in females was stated by NTP (2020) to be consistent with the lower plasma PFOA

1 levels due to the rapid excretion of PFOA in female rats. The 2020 NTP study identified clear
2 evidence of carcinogenic activity in the liver and pancreas in rats exposed over a lifetime,
3 including the pre-natal, pre-weaning, and post-weaning stages. While the NTP carcinogenicity
4 studies do not include conclusions about human carcinogenicity, the 2020 NTP study
5 concluded “clear evidence” of carcinogenic activity in male rats, which is the strongest level of
6 evidence category used by the NTP in evaluating data from studies of this kind.

7 Mode of action analyses (PFOA HESD, 2016; DWQI, 2017) have concluded that the rat liver
8 tumors caused by PFOA may not be relevant to humans, although this is not a settled issue.
9 However, the mode of action for other types of tumors caused by PFOA in rats has not been
10 established, and as specified in the U.S. EPA (2005) *Guidelines for Carcinogen Risk Assessment*,
11 they are considered relevant to humans.

12 An additional study that is relevant to the mode of action for carcinogenicity of PFOA is the
13 initiation-promotion study in rainbow trout (Benninghoff *et al.*, 2012). In this study, PFOA
14 significantly increased the number of tumors and the diameter of liver tumors compared to
15 controls treated with the same initiator. The increase in number of liver tumors was greater for
16 PFOA than for PFOS, and PFOS did not cause a significant increase in tumor diameter. As
17 reviewed in DWQI (2017), the overall significance of this study is that PFOA increased hepatic
18 tumors in "rainbow trout, a species used as a model for human liver carcinogenesis because it is
19 insensitive to peroxisome proliferation [e.g., PPAR- α activation], suggest[ing] that PFOA
20 promotes liver tumor development through an estrogenic mechanism."

21 Finally, the draft PFOA document (p. 343-344) discusses a genotoxic versus non-genotoxic
22 mode of action for carcinogenicity of PFOA, followed by the conclusion that PFOA is
23 considered "Likely to Be Carcinogenic to Humans." As such, it should be made clear that the
24 designation of "likely to be carcinogenic to humans" is independent of whether the mode of
25 action is genotoxic or non-genotoxic.

26 *Rationale for “likely carcinogen” designation*

27 The designation of PFOA as "likely to be carcinogenic to humans" would have large practical
28 implications because the MCLG for "likely carcinogens" is zero. For this reason, it is particularly
29 important for EPA to provide a strong and transparent rationale for reaching this conclusion. It
30 would enhance transparency and confidence to have an objective, well-described, systematic
31 approach for evidence synthesis that incorporates the prior studies included in the 2016 HESD
32 and the newer studies. In fact, this “single integrative step” of weighing all the evidence after
33 assessing the individual lines of evidence is emphasized in the U.S. EPA (2005) *Guidelines for*
34 *Carcinogen Risk Assessment*.

35 As discussed above, human, animal, and mode of action studies support the designation of PFOA
36 as “likely to be carcinogenic to humans.” However, the rationale for this designation is not
37 adequately provided in the draft MCLG document. Additional “weight of evidence narrative” is
38 needed, including discussion of epidemiologic data structured along the Hill criteria, assessment
39 of evidence from animal studies, and assessment of mode of action, that justifies why the
40 evidence fits into the “likely” “descriptor.

The EPA *Guidelines for Carcinogen Risk Assessment* emphasize the importance of weighing all of the evidence in reaching conclusions about the human carcinogenic potential of agents. Specifically, the *Guidelines for Carcinogen Risk Assessment* provide at least five examples of data that support the “likely” descriptor (listed above). The draft MCLG document should explicitly demonstrate how the available data for PFOA are consistent with one or more of those examples in order to fulfill the criteria for designation as a “likely” carcinogen. These *Guidelines* are cited but the specific examples of supporting data should be added to the draft document.

Finally, the weight of evidence evaluation (i.e., determination of the appropriate descriptor for carcinogenic potential, such as “suggestive” or “likely”) is part of the Hazard Identification component of the risk assessment, not the Dose-Response component. As such, the Weight of Evidence section (section 4.2) should be moved from the Dose-Response section to the Hazard Identification section (e.g., Evidence Integration) on cancer.

Consideration of other designations

The Panel agreed that neither of the categories of “not likely to be carcinogenic to humans” or “inadequate information to assess carcinogenic potential” are appropriate, as epidemiologic studies and studies of experimental animal models/chronic cancer bioassays do exist and cancers/tumors have been observed in relation to PFOA exposure. There is also agreement that the data exceed the descriptor for “suggestive evidence of carcinogenicity,” which may include a positive cancer result from only a single animal or human study with additional studies of mixed results.

The Panel also considered whether the currently available data for PFOA may exceed the descriptor for “likely” evidence and meet the criteria for the higher designation of “carcinogenic to humans”. This descriptor of “carcinogenic to humans” indicates strong evidence of human carcinogenicity. According to the *Guidelines for Carcinogen Risk Assessment*:

- This descriptor is appropriate when there is convincing epidemiologic evidence of a causal association between human exposure and cancer.
- Exceptionally, this descriptor may be equally appropriate with a lesser weight of epidemiologic evidence that is strengthened by other lines of evidence. It can be used when all of the following conditions are met: (a) there is strong evidence of an association between human exposure and either cancer or the key precursor events of the agent's mode of action but not enough for a causal association, and (b) there is extensive evidence of carcinogenicity in animals, and (c) the mode(s) of carcinogenic action and associated key precursor events have been identified in animals, and (d) there is strong evidence that the key precursor events that precede the cancer response in animals are anticipated to occur in humans and progress to tumors, based on available biological information.

The Panel concluded that the available epidemiologic data do not provide convincing evidence of a causal association but rather provide evidence of a plausible association, and thus do not support a higher designation of “carcinogenic to humans.” While Bartell and Vieira (2021), in a critical review and meta-analysis of epidemiological literature concluded that associations

between PFOA and kidney and testicular cancer were “likely causal”, they noted the limited number of studies and the need for additional larger cohort studies to support this conclusion. As discussed above, the epidemiologic data supporting the designation of “likely carcinogenic” derive largely from several *medium confidence* studies (including Shearer *et al.* 2021) and in the absence of any *high confidence* epidemiologic studies. The Shearer *et al.* (2021) showed an increased risk of kidney cancer in the highest PFOA exposure quartile (odds ratio, OR =2.63) that was slightly attenuated (OR=2.19) and not statistically significant after adjusting for other PFAS. Thus, the Shearer *et al.* (2021) study should not be overemphasized, as “an inference of causality is strengthened when a pattern of elevated risks is observed across several independent studies. The reproducibility of findings constitutes one of the strongest arguments for causality” (U.S. EPA *Guidelines for Carcinogen Risk Assessment*, 2005).

The Panel agreed that the EPA document again needs additional discussion of the “weight of the evidence” that supports the exclusion of the higher level of designation as “carcinogenic”. The draft document should explicitly demonstrate how the available data for PFOA do not meet one or more of the examples of the criteria for designation as a “carcinogen.”

Finally, the Panel noted that the practical impact of a “likely to be carcinogenic” and a “carcinogenic” designation on the MCLG are the same, since the MCLG would be set at zero in either case.

Recommendations

While the Panel agrees with the “likely” designation for PFOA carcinogenicity based on new evidence and prior evidence included in the 2016 HESD, there is a need for a more structured and transparent “weight of evidence” discussion to support the rationale behind this designation, including:

- explicit description of how the available data for PFOA are consistent with one or more of the criteria in the EPA *Guidelines for Carcinogen Risk Assessment* (2005) for designation as a “likely” carcinogen;
- explicit description of how the available data for PFOA do not meet the criteria for the higher designation as “carcinogenic.”

B. PFOS: Based on a small number of new cancer studies identified since the 2016 PFOS HA, EPA concludes that the available cancer data for PFOS indicate a ‘suggestive’ categorization which is unchanged from the categorization identified in the 2016 HA. Does the panel agree that the new studies do not change the designation? If yes, is the rationale clearly described? If no, please provide an explanation for arriving at a different conclusion.

Designation of PFOS as “suggestive evidence of carcinogenic potential”

As described above in the response to Charge Question 3a concerning the cancer classification for PFOA, EPA *Guidelines for Carcinogen Risk Assessment* (2005) provide a structured approach for assessing the weight of evidence regarding carcinogenic potential of an agent and for designation as: carcinogenic to humans, likely to be carcinogenic to humans, suggestive evidence of carcinogenic potential, inadequate information to assess carcinogenic potential, or not likely to be carcinogenic to humans.

Based on EPA's *Guidelines for Carcinogen Risk Assessment* (2005), supporting data for the "suggestive" descriptor may include:

- a small, and possibly not statistically significant, increase in tumor incidence observed in a single animal or human study that does not reach the weight of evidence for the descriptor "Likely to Be Carcinogenic to Humans." The study generally would not be contradicted by other studies of equal quality in the same population group or experimental system;
- a small increase in a tumor with a high background rate in that sex and strain, when there is some but insufficient evidence that the observed tumors may be due to intrinsic factors that cause background tumors and not due to the agent being assessed. (When there is a high background rate of a specific tumor in animals of a particular sex and strain, then there may be biological factors operating independently of the agent being assessed that could be responsible for the development of the observed tumors.) In this case, the reasons for determining that the tumors are not due to the agent are explained;
- evidence of a positive response in a study whose power, design, or conduct limits the ability to draw a confident conclusion (but does not make the study fatally flawed), but where the carcinogenic potential is strengthened by other lines of evidence (such as structure-activity relationships); or
- a statistically significant increase at one dose only, but no significant response at the other doses and no overall trend.

Based on the EPA *Guidelines*, the Panel discussed whether the available data support a designation of "suggestive" which is unchanged since the 2016 categorization; or meet the higher descriptor for "likely to be carcinogenic to humans". Several new studies have been published that warrant further evaluation to determine whether the "likely" designation is appropriate. Below, the Panel provides some information that the EPA should consider in the overall weight of evidence for carcinogenicity of PFOS.

Supporting epidemiologic and experimental evidence for "suggestive" designation

The epidemiological evidence for the carcinogenicity of PFOS is mixed and/or methodological limitations of the studies preclude firm conclusions. Of the 11 new studies identified since 2016, 8 were considered medium confidence and 3 were considered low confidence. There were no new animal toxicity studies identified. A single chronic cancer bioassay in rodents (Butenhoff *et al.*, 2012) showed increases in tumors in the liver, thyroid gland, and mammary gland; these tumors did not appear in a dose-responsive pattern.

In the draft EPA PFOS document, the Hazard Identification section on cancer (Section 3.3.1.7) discusses that Shearer *et al.* (2021) study showing an association of PFOS with kidney cancer. Comparison of the discussion of this topic in Section 3.3.1.7 of the PFOS document with the discussion of the same topic in the PFOA document indicates that some, but not all, of the specific analyses that were statistically significant for PFOA were not significant for PFOS.

Specifically, the draft EPA PFOS document states (p. 286-287): "PFOS was associated with an increased risk of kidney cancer (i.e., renal cell carcinoma) in a medium confidence study {Shearer, 2021, 7161466}. The study reported a statistically significant increase in risk in the highest exposure quartile and per doubling of PFOS concentration. After adjusting for other PFAS the association remained elevated in the highest quartile (i.e., adjusted OR=1.14), but it was no longer statistically significant and was lower than the second quartile; additionally, there was no association when evaluated on a per doubling of PFOS."

This can be compared to the information from the draft EPA PFOA document (p. 309): "PFOA was associated with an increased risk of kidney cancer (i.e., renal cell carcinoma (RCC)) {Shearer, 2021, 7161466}. This large medium confidence case-control study nested within the NCI's Prostate, Lung, Colorectal, and Ovarian Screening Trial (PLCO) reported a statistically significant increase in risk of kidney cancer in highest exposure quartile and per doubling of PFOA concentration. Even after adjusting for other PFAS the association remained significant in analyses on a per doubling increase in PFOA. The increase in the highest exposure quartile remained and the magnitude was similar (i.e., OR = 2.63 without adjusting for other PFAS vs. 2.19 after adjusting for other PFAS), but it was no longer statistically significant."

The Panel noted that draft PFOS document does not mention that the Shearer *et al.* (2021) results were used for a CSF for PFOA. Thus, further discussion of the Shearer *et al.* (2021) epidemiologic study findings (classified as a *medium confidence* study) as they pertain to PFOS are needed, including the weight of evidence section (Section 4.2.1) that does not mention Shearer *et al.* (2021) at all in its discussion of new PFOS cancer studies identified since the 2016 HESD. In particular, the Panel concluded that the PFOS document should clearly indicate why the findings of the Shearer *et al.* (2021) study were judged to be less definitive for PFOS than for PFOA. The magnitude of the association between PFOS and kidney cancer was lower than that for PFOA, and after adjustment for other PFAS the adjusted OR for the highest quartile was 1.14 and not statistically significant. However, these data should be presented clearly including a discussion of why the PFOS data from Shearer *et al.* (2021) were not considered sufficient for a higher designation of "likely carcinogenic."

The Panel also suggests that Li *et al.* (2022), a recent study that may be relevant to weight of evidence for carcinogenicity of PFOS, be included in the discussion. This study reported an increased incidence of kidney cancer in a Swedish population exposed to drinking water contaminated with a mixture of PFAS, with PFOS predominating.

An additional study that is relevant to the mode of action for carcinogenicity of PFOS is the initiation-promotion study in rainbow trout (Benninghoff *et al.*, 2012). This study was not included in the PFOS document. In this study, PFOS significantly increased the number of tumors compared to controls treated with the same initiator; this increase was greater for PFOA

than for PFOS. PFOS did not cause a significant increase in tumor diameter compared to controls treated with the same initiator, while this effect was significant for PFOA. As reviewed in DWQI (2017), the overall significance of this study is that PFAS increased hepatic tumors in "rainbow trout, a species used as a model for human liver carcinogenesis because it is insensitive to peroxisome proliferation [e.g., PPAR- α activation], suggest[ing] that PFOA promotes liver tumor development through an estrogenic mechanism." As discussed previously, this study is important to understanding the mode of action of hepatic tumors caused by PFOA because it has been suggested that rodent liver tumors caused by PFOA occur through PPAR- α activation that is not relevant to humans. However, this is not an issue for PFOS because the rodent liver tumors caused by PFOS do not appear to be PPAR- α dependent. For example, DWQI (2018) concluded that "available data do not support the conclusion that PFOS causes liver effects through a PPAR α -dependent mode of action at the doses that resulted in tumors in [the chronic rat study conducted by] Butenhoff *et al.*, 2012."

The Panel suggests mammary tumor development be considered as an additional endpoint associated with PFOS. Human (Bonefeld-Jorgensen *et al.*, 2011; Wielsøe *et al.*, 2017; Cohn *et al.*, 2020; Mancini *et al.*, 2019), animal (Butenhoff *et al.*, 2012), and mechanistic (Pierozan *et al.*, 2018) studies support this as an endpoint of concern, warranting a POD derivation. For mammary tumor development, there are some recent studies that have shown epidemiologic and mechanistic evidence linking PFOS with breast cancer outcomes. On the epidemiologic side, Mancini *et al.*, 2019 found a linear dose response for PFOS and receptor positive tumors (there was also a potential association with ER- tumors). On the mechanistic side, Pierozan *et al.*, 2018 showed that exposure to PFOS (at 10 microM) induced malignant transformation of MCF-10A cells.

Rationale for "suggestive" designation

Again, the findings of the Shearer *et al.* (2021) study for PFOS need to be presented clearly including a discussion of why they were not considered sufficient for a higher designation of "likely carcinogenic."

The Panel also recommend additional discussion to support the rationale for the "suggestive" designation as opposed to the "likely" designation be presented, and this recommendation stems largely from lack of inclusion of mechanistic data and inconsistency with the California EPA conclusions based on the same human, animal, and mechanistic evidence presented in the EPA PFOS document. The Panel noted that the New Jersey DWQI (2018) determined that draft PFOS should be described as having suggestive evidence of carcinogenicity, using EPA (2005) *Guidelines for Carcinogen Risk Assessment* and consistent with the designation in the draft EPA PFOS document, while observing that this conclusion was made prior to the publication of Shearer *et al.* (2021).

The criteria used by California EPA for determination that a chemical is a carcinogen are not identical to the criteria in the U.S. EPA (2005) *Guidelines for Carcinogen Risk Assessment*. California has proposed (effective December 24, 2021) listing PFOS and its salts and transformation and degradation precursors on the list of chemicals known to the state to cause cancer for purposes of the Safe Water Drinking and Toxic Enforcement Act of 1986 (Proposition

65) (OEHHA, 2021a). The OEHHA summary of human evidence indicates that the results were mixed, and the summary of animal evidence highlights a chronic carcinogenicity study in rodents supported by the tumor promotion study in rainbow trout (OEHHA, 2021b), which is mostly consistent with evidence provided in the PFOS document.

The mechanistic data in the OEHHA summary (OEHHA, 2021b) were used to identify the shared characteristics between PFOS and other known carcinogens. Of particular importance in the OEHHA document was the finding of suggestive evidence that PFOS (and PFOA) is genotoxic – leaving the potential for a genotoxic MOA that differs from the 2016 HESD for PFOS. Given that the EPA has not completed the review of mechanistic studies for the cancer endpoint, it seems premature to have kept the 2016 determination without further evaluation of the mechanistic evidence.

In addition to the lack of inclusion of mechanistic data in the weight of evidence for PFOS carcinogenicity, the interpretation of the hepatocellular carcinoma data from the Butenhoff (2012) study in the 2016 HESD is overly conservative in dismissing the appearance of a dose-response relationship for this endpoint. Relevant to this point, it is noted that DWQI (2018) developed a CSF based on the incidence of hepatocellular tumors in Butenhoff *et al.* (2012). Given that multiple MOAs may be operative in this outcome, the Panel suggests a reevaluation of the 2012 Butenhoff study is warranted.

Finally, as mentioned for PFOA above, the weight of evidence evaluation (i.e., determination of the appropriate descriptor for carcinogenic potential, such as "suggestive" or "likely") is part of the Hazard Identification component of the risk assessment, not the Dose-Response component. As such, the Weight of Evidence section should be moved from the Dose-Response section to the Hazard Identification section (e.g., Evidence Integration) for cancer.

Recommendations

The Panel recommends that a more structured and transparent “weight of evidence” discussion be added. Specific areas that should be addressed include:

- explicit description of why the available data for PFOS do not meet the EPA *Guidelines for Carcinogen Risk Assessment* (2005) criterion for the higher designation as “likely carcinogenic”
- inclusion and discussion of mechanistic data

The Panel also recommends that the findings of the Shearer et al (2021) study for PFOS be presented clearly including a discussion of why they were judged to be less definitive for PFOS than for PFOA and not considered sufficient to support a higher designation of “likely carcinogenic.”

Charge Question #3 – Cancer Slope Quantification

B. *Cancer Slope Quantification: EPA used the Shearer et al., 2021 epidemiological study to quantify a cancer slope factor using peak exposure for PFOA. Has EPA adequately justified the use of this study and peak exposure for the quantification of a cancer slope factor for PFOA? If no, please describe alternate approaches that SAB recommends. Does SAB support the selection of this CSF in the derivation of a risk specific dose for PFOA (i.e., the concentration of PFOA in drinking water that would have a one-in-1-million chance of an increased cancer risk)? If not, please provide input on the strengths and weaknesses of the other candidate CSFs that EPA derived.*

At the SAB Panel meeting on December 16, 2021, EPA clarified that the term "peak exposure" should be disregarded in the charge question, and that input is requested on the development of the CSF in general, and risk specific dose for PFOA.

In general, the Panel agrees that it is preferable to base the CSF derivation on human epidemiological data when appropriate human data are available. However, for PFOA, there is an absence of “high confidence” epidemiologic data as summarized by EPA.

EPA selected the Shearer *et al.*, 2021 study for CSF derivation. The systematic review conducted by EPA categorized the overall confidence in this study as “medium”, due to a deficiency in controlling for confounding, and adequate confidence in selectivity and sensitivity of the study (according to Figure 123, though not specifically described in the text). The Panel agreed with this classification, noting some merits and several limitations of the study design and overall significance of the results.

The Shearer *et al.*, 2021 study is a prospective epidemiologic study that investigated the association between exposure to eight PFAS, including PFOA, and RCC risk, with PFAS measurements for all subjects in serum collected prior to RCC diagnosis. This study adjusted for diminished kidney function in order to control for reverse causation due to reduced kidney function in the observed associations. The study authors reported that, after adjusting for PFOS and other PFAS, the increased risk of kidney cancer in the highest PFOA exposure quartile compared to the lowest was attenuated (from OR=2.63 to OR=2.19), and neither that effect estimate for the highest quartile, nor the dose-response trend remained statistically significant. However, when PFOA was modeled continuously, the association remained statistically significant after adjusting for other PFAS. Although not discussed in the paper, the Panel noted that Supplementary Figure 1 shows that one individual in the RCC group had much higher serum PFOA than any of the other cases or controls. The impact of this individual on the elevated RCC risk in the highest exposure group is unclear. The Panel suggests that EPA investigate this question, including contacting the authors of Shearer *et al.* (2021) if appropriate.

The study also revealed associations with RCC for PFOS, and PFHxS, which were reported to be moderately correlated with PFOA, in models unadjusted for other PFAS. Based on this finding, it is not clear whether other PFAS (e.g., PFNA) that were also moderately correlated with PFOA were adequately controlled for. The issue of impacts of other correlated PFAS on the apparent dose-response for a specific PFAS is discussed in the response to the earlier charge question #2

on confounding in studies of noncancer effects, and this issue should be included in the discussion of Shearer *et al.* (2021) in the draft MCLG documents.

Overall, the epidemiological studies have not consistently identified associations between PFOA and RCC; some epidemiological studies support RCC as a critical finding associated with PFOA exposure while others (with several limitations noted) have failed to detect an association between PFOA and RCC. Concerning the human studies that have shown an association of PFOA with cancer (Shearer *et al.* (2021), Vieira *et al.* (2013), Barry *et al.* (2013), and Steenland and Woskie (2012)), the draft MCLG document did not provide the rationale for using Shearer *et al.* (2021) rather than of one of the other studies to derive a CSF for PFOA. In contrast, California EPA (2021) does provide a rationale for using Shearer *et al.* (2021) to derive a CSF. Further, California EPA's (2021) approach for deriving a CSF for PFOA and kidney cancer was based on serum PFOA levels (ng/ml)⁻¹ from a study of the general population (Shearer *et al.*, 2021) and a study of communities exposed to PFOA in drinking water (Vieira *et al.*, 2013), with the recommended draft CSF (CalEPA, 2021) proposed as the average of the two CSFs. Given that the Panel has identified several limitations of the Shearer (2021) study, and the EPA classification of this study is of "medium" overall confidence, a clear rationale for its selection as the sole basis for the CSF should be provided.

The 8 epidemiologic studies identified in the EPA's systematic review were all considered of "medium" overall confidence, whereas the NTP 2020 chronic bioassay in rats is considered a "high" confidence study. However, a CSF was derived for only one of the epidemiologic studies. The Panel agrees that toxicity values should only be derived from studies with at least "medium" confidence, but the draft document needs to be more transparent as to weighing the strengths and limitations of different studies to support a CSF (including both human and animal studies).

Some human studies were excluded during the systematic review, including those of "special populations" and "occupational" exposure studies. While PFOA exposures might be higher in these populations relative to the general population, these studies could be complementary, and for consistency and transparency, should be similarly evaluated as the other studies included in the draft document. The rationale for their inclusion or exclusion in the development of candidate CSFs should be presented.

The NTP (2020) study reported increased incidences of hepatocellular adenomas (or carcinomas) and PACTs. Of the three chronic bioassays that have been conducted in rodents (NTP 2020, Biegel *et al.*, 2001, Butenhoff *et al.*, 2012), none observed an increased incidence in kidney cancers. While concordance of tumor sites between animal and human exposures is not always observed and the EPA (2005) Guidelines for Carcinogen Risk Assessment state that "target organ concordance is not a prerequisite for evaluating the implications of animal study results for humans," the lack of "high" confidence epidemiological studies would support that "high" confidence animal studies might be at least of equal weight to "medium confidence" human studies for CSF derivation.

Overall, the Panel recommends that multiple candidate CSFs be developed, including those based on additional epidemiologic studies of sufficient quality (which is important given the exclusion of earlier human studies from consideration in the draft document) as well as animal

1 cancer bioassays. Each study's strengths and limitations (e.g., exposure uncertainties,
2 confounding [including possible reverse causation], statistical power) should be discussed and
3 then a judgment made as to whether to select one or more studies to represent the overall slope
4 factor. The Panel does note that the CSFs derived from the Shearer *et al.* (2021) study appear to
5 be two to three orders of magnitude more potent than those derived from experimental animal
6 studies, and thus the decision as to what slope factor to recommend needs to be carefully
7 considered and highly transparent.

8 The Panel also identified other aspects of the CSF and risk specific dose for PFOA that should be
9 addressed and/or further discussed in the document. They are listed below.

10 Lifetime RCC risk estimation relied on lifetime risk of kidney cancer in US males. RCC
11 baseline risk is higher in Black individuals compared to White individuals. Discussion
12 and/or demonstration of how the CSF might differ if different rates were applied that are
13 representative of the population should be included.

14 Multiple statements in the draft MCLG documents indicated that MOA results across PFOA
15 and PFOS are currently under review and that EPA's MOA evaluation would be deferred
16 until after the SAB review. The Panel commented that including the final MOA evaluation
17 for kidney cancer is important if this endpoint is used by EPA.

18 The draft PFOA document states that the CSF for Shearer *et al.* (2021) is based on the modeling
19 approach used by California EPA (2021), and in the corrected version of Table 25 in the draft
20 PFOA document is the California EPA (2021) CSF based on the central tendency estimate serum
21 level from Shearer *et al.* (2021). At the SAB meeting on December 16, 2021, EPA stated that
22 they had independently replicated the modeling performed by California EPA. However, the
23 details of the modeling and its results, as well as, the conclusions of the EPA review, are not
24 shown in the draft PFOA document. The Panel recommends that this information be included in
25 the final EPA document, as well as, further discussion and clarification as described below.

26 The draft PFOA document (Table 25) shows the CSF from Shearer *et al.* (2021) as administered
27 dose (ng/kg/day)⁻¹ as well as serum PFOA (ng/ml)⁻¹. However, California EPA (2021) does not
28 provide the CSF in terms of administered dose. EPA's derivation of the CSF as administered
29 dose (ng/kg/day)⁻¹ from the PFOA serum level (ng/ml)⁻¹ CSF is not discussed in the main
30 document or appendices (Appendix B.1.5.1 briefly discusses the calculation of the CSF from
31 Shearer *et al.* (2021)). As the clearance factor was not mentioned or provided in the document,
32 one reviewer calculated that the serum level CSF was converted to the administered dose CSF
33 using a clearance factor of 0.12 ml/kg/day, based on a half-life of 2.7 years and a volume of
34 distribution of 170 ml/kg (values stated to have been selected by EPA in Sections 3.2.3 and 3.2.4
35 of the document). Applying this clearance factor to the serum level CSFs (Table 25) would result
36 in the administered dose CSFs that are shown. Additionally, although the numerical value of the
37 central tendency slope factor (0.00178) is correctly shown in Appendix B.1.5.1, the units shown,
38 (ng/kg/day)⁻¹ instead of (ng/ml)⁻¹, are incorrect. The development of the clearance factor and its
39 use in determining that administered dose CSFs from the serum level CSFs should be clearly and
40 completely described in the final document.

Table 25 of the draft PFOA document also shows the 95% upper confidence level of the CSF from Shearer *et al.* (2021). However, the 95% upper confidence level CSF is not provided in California EPA (2021), and no information on the modeling results that provided this value are shown in the draft MCLG document; these modeling results should be provided. It is also not clear from the information in the draft PFOA document whether the central tendency CSF or the upper 95th percentile CSF would be used to develop a risk specific dose, and this should be clarified.

Recommendations

The Panel recommends that multiple candidate CSFs be developed, including those based on additional epidemiologic studies of sufficient quality as well as animal cancer bioassays. Strengths and limitations for each study should need to be discussed and a judgment made as to whether to select one or more studies to represent the overall slope factor.

The Panel recommends that a MOA evaluation for kidney cancer be included if this endpoint is used by EPA.

The Panel recommends that the details of the modeling and its results for the derivation of a CSF from Shearer *et al.* (2021), and the conclusions of the EPA review of this information that are not shown in the draft PFOA document be included in the final EPA document.

The Panel recommends that the development of the clearance factor and its use in determining that administered dose CSFs from the serum level CSFs (from Shearer (2021) be clearly and completely described in the final document.

Toxicokinetic Modeling – General SAB Comments

The transparency of the toxicokinetic (TK) modeling can be substantially strengthened by improving the documentation of the models and their applications in both humans and animals. The Panel found it challenging to locate the information necessary to piece together how the models were used, as much more information is needed beyond citations and tables of model parameters. As an example, the BMD calculations in the Appendix apparently use model generated blood concentrations for dose-response analyses, but the specific details are absent. *Example of need for additional information as to what models/approaches were used for HED derivation*

The Panel noted a need for additional transparency; a critical example of this is in development of the PODHED for decreased vaccine response when calculating “the dose to mothers & children that results in the same serum concentration at 5 years of age.” Specifically, in the draft PFOA document, it is not clear from the information provided in Section 4.1.3.2 (Toxicokinetic Model for Human Dosimetry) that prenatal and breastfeeding exposures were considered in development of the PODHED for decreased vaccine response from exposure at age 5.

Information on this topic is especially important because these PODs are the basis for the final PFOA and PFOS Reference Doses in the draft MCLG documents.

This is confusing, particularly because the excerpt from Section 4.1.3.2 of the draft PFOA document copied below (emphasis added) indicates that the steady-state assumption was used to develop the PODHED for all of the human endpoints (including the PODHED based on exposure at age 5), and that exposures during early life-stages were only considered in animal developmental studies.

“This updated [Verner et al., 2016] model was used to simulate the HED from the animal PODs that were obtained from BMD modeling of the animal studies (Section B.2). It was also used to simulate selected human studies to obtain a chronic dose that would result in the internal POD obtained from dose-response modeling. For PODs resulting from chronic exposure, such as a long-term animal study or a human study, the steady state approximation was used to calculate a HED that would result in the same dose metric after chronic exposure. For PODs from exposure to developmental animals, the life-stage developmental model was used to calculate a HED that results in the same dose metric during the developmental window selected by the Dose-Response team.”

By contrast, in the draft PFOS document, it is indicated that prenatal and breastfeeding exposure was considered in development of the PODHED for decreased vaccine response from exposure at age 5, and EPA confirmed this at the Panel meeting on December 16, 2021. However, this topic is not discussed in the text; it is only mentioned in a footnote to the PODHED for decreased antibody response to tetanus and diphtheria vaccines in Table 21 as follows: “Calculated as the dose to mothers & children that results in the same serum concentration at 5 years of age. Note that the model predicted slightly different serum concentrations for male and female children, so the lower HED was selected to be more health protective.”

These presence of such discrepancies and ambiguities necessitate clearer and more transparent documentation as to the specific models, model parameters, and other simulation inputs that are used for each HED derivation, as recommended below.

Recommendations

The Panel recommends that Model performance, along with a statement as to acceptable performance metrics, should be documented for every model (including for different life stages). For instance, plotting predicted and observed concentrations as scatter plot can be helpful to evaluate overall bias and precision (e.g., including lines at 1, 2x, and 0.5x to put performance in context, with portion of samples outside these bounds giving some indication of the acceptability of the model). Comparisons of data and time-course simulations can be helpful as well. If there are no data available for evaluating performance, this can be stated for a particular life stage.

The Panel recommends that when a model is used in dose-response analyses, the details and assumptions need to be documented sufficiently so that someone can reproduce the simulations, as noted above. Specifically, for every human or animal simulation there should be information stating which model was employed and what model parameter and input values were used to

1 simulate the specific study or scenario, with the code made available so someone can reproduce
2 the work. It may be helpful to develop a “big picture” workflow schematic for the TK model,
3 how they fit into the BMD and human equivalent dose (HED) calculations.

4 The Panel recommends that EPA should better characterize the uncertainty that results from
5 different parameters/ assumptions, such as through sensitivity analyses or Monte Carlo
6 simulation with a range or distribution of values. For instance, the Goeden *et al.* (2019)
7 transgenerational TK model includes at least both central and upper bound estimates for different
8 parameters which could serve as the basis for a sensitivity analysis.

9 The Panel recommends that although the draft MCLG documents develop Reference Doses, not
10 MCLGs, EPA should develop a Reference Dose based on serum PFOA levels that can be used to
11 develop a drinking water concentration (MCLG) that is protective for all life-stages. Specifically,
12 such model results would be used along with life-stage-specific changes in ingestion rates at a
13 fixed water concentration in order to be the basis of an MCLG.

14 Charge Question #4 - Human Toxicokinetic Model

15 *A. For endpoints observed in adults, EPA used a steady-state approach to calculate the*
16 *HED, which assumes a relatively constant exposure and clearance during adulthood. Please*
17 *comment on this method of HED calculation. Are there alternative approaches that EPA should*
18 *consider? If so, please describe the rationale for recommending this approach(es).*

19
20 *Two key parameters are the half-life and volume of distribution, which were used to calculate*
21 *clearance. Half-life and volume of distribution were assumed to be constant across sex and age*
22 *groups because of a lack of strong quantitative data to parametrize changes across sex and age.*
23 *Please comment on the strengths and weakness of the use of this assumption and the choice of*
24 *these parameters by the EPA. Please describe the rationale for alternative recommended*
25 *approaches.*

26 In general, the Panel agreed that for adults and chronic exposure, the PFOA and PFOS
27 compartmental models for adults (not pregnant or lactating females) are adequate for use in HED
28 determinations. There is also general agreement that the assumption of a constant half-life and
29 volume of distribution for human adults is reasonable. Additionally, the assumptions of steady
30 state are reasonable, given the long half-life of each molecule. The Panel notes that this approach
31 is highly empirical and limited when looking to the future and using this model to ask questions
32 about mixtures.

33 However, as noted in the overall comments, more details as to the model code, parameters, data,
34 and performance are needed in order to transparently evaluate the model, and to justify the
35 selection of parameters. While Appendix D reviews the many different estimates, a methodology
36 for picking a value for a particular case to use was not clearly described. For instance, Section
37 4.1.3.2 of the draft MCLG documents discuss the parameters from the Verner *et al.* (2016)
38 model that were modified by EPA, and these do not include the volume of distribution or half-
39 life. For both PFOA and PFOS, the volume of distribution used in Verner *et al.* (2016) is
40 identical to the value stated to have been selected by EPA in Section 3.2.4 of the draft MCLG
41 documents (0.17 L/kg for PFOA; 0.23 L/kg for PFOS). However, the half-lives of 3.8 years for

PFOA and 5.4 years for PFOS used by Verner *et al.* (2016) differ from the half-lives of 2.7 years for PFOA and 3.8 years for PFOS stated to have been selected by EPA in Sections 3.2.3 of the draft MCLG documents. It is unclear whether EPA used the half-life values that it selected or those selected by Verner *et al.* (2016) when applying the Verner *et al.* (2016) model. Also, if EPA did not use the half-life values it selected when applying the Verner *et al.* (2016) model, it is unclear where they were actually used in the EPA evaluations. This information should be clarified.

Recommendations

The Panel recommends that more details as to the model code, parameters, data, and performance be included in order to provide transparent evaluation the model and justification the selection of parameters.

B. For endpoints observed in human neonates or children, EPA used a one-compartment TK model to simulate dosimetry during pregnancy and a two-compartment TK model (one-compartment models for the mother and the child) to simulate dosimetry during lactation, to calculate the HED for each POD. Please comment on the strengths and weaknesses of this choice of model structure for the task of predicting dosimetry in the human fetus and child compared to dosimetry in mice and rats in the similar lifestages. Please provide the rationale for any alternative recommended approaches.

Overall approach for human neonates or children

The Panel agreed that compartmental models (rather than the more complex physiologically-based pharmacokinetic [PBPK] models) are reasonable for simulating dosimetry during pregnancy and lactation for endpoints in human neonates and children for PFOA and PFOS. The strengths of using this compartmental modeling approach include that it provides a data driven estimate pharmacokinetics, it avoids numerous complexities, and it is less time consuming. However, a disadvantage is that this methodology might not be as generalizable because it is highly empirical, and does not allow for the future use of mechanistic information and scaling methods, addressing mixtures, or PFAS with different pharmacokinetic (PK) behaviors and limited or no PK data. Although the PK behavior of PFOS and PFOA should be straight forward given available data, the mechanisms governing the PK behavior are not simple. Protein binding and active transport of PFOS and PFOA into and out tissues by protein transporters occur, to such a degree that the pharmacokinetics is altered. What this looks like in a growing fetus and neonate/infant is not well understood based on PFOA or PFOS longitudinal data.

In light of these disadvantages, future consideration (not for this case of PFOA and PFOS MCLG development) should be given to utilizing the broader foundation of life-stage PBPK modeling (see **Box 1**). Specifically, there is a wealth of information now for pediatric drugs and PBPK model that provides a foundational approach for addressing life-stage PBPK modeling in humans. There are literally hundreds of publications available in the literature (see example references in **Box 1**). The strategy is to construct a modeling framework that can be used for

Box 1: Illustration of life-stage modeling principles

This is a brief example of some of the principles of life-stage modeling for future consideration. The Panel is not recommending this course of action now for PFOA and PFOS, but is using PFOA as an illustrative example. The PFOA pKa and the pH of plasma (and other tissue groups) would suggest that a small portion of the circulating PFPA is unionized, and the rest is ionized, bound and unbound to serum proteins. The unionized fraction is available for passive transport across membranes and the ionized fraction, by protein transporters. Urinary clearance (CL_u) would be described with glomerular filtration rate (GFR, passive), secretion, and reabsorption. With PFOA, if CL_u in mice, rat, or human was less than GFR, this would signal further evaluation and consideration of reabsorption by kidney protein transporters. The ontogeny of protein transporters (e.g., kidney, liver) is an active field of research and can be used for extrapolation or interpolation based on protein abundance or activity to provide simulations without the benefit of data. Ontogenies of important serum proteins are known and would be useful for describing PFOA binding. This approach is very different from the current methods used by EPA for PFOA and PFOS to address life stages. Ultimately this methodology provides a biologically based anchoring of key parameters that can be used for understanding the pharmacokinetics of chemicals across life stages and reproductive states. Of course, there can still be gaps in data or knowledge. However, if time is invested in exploring the modeling work completed over the last 11 years for pediatric drugs, and for the period during the last 3-4 years for pregnancy and lactation, many ideas are likely to emerge.

Selected References

- Cristea *et al.* 2020, The Influence of Drug Properties and Ontogeny of Transporters on Pediatric Renal Clearance through Glomerular Filtration and Active Secretion: A Simulation-Based Study.
- Huh *et al.* 2011, Interspecies scaling and prediction of human clearance: comparison of small-and macro-molecule drugs
- Johnson and Ke 2021, Physiologically Based Pharmacokinetic Modeling and Allometric Scaling in Pediatric Drug Development: Where Do We Draw the Line?
- Mahmood 1999, "Prediction of Clearance, Volume of Distribution and Half-life by Allometric Scaling and by use of Plasma Concentrations Predicted from Pharmacokinetic Constants.
- Paine *et al.* 2011, Prediction of Human Renal Clearance from Preclinical Species for a Diverse Set of Drugs That Exhibit Both Active Secretion and Net Reabsorption
- Van Groen *et al.* 2021, Ontogeny of Hepatic Transporters and Drug Metabolizing Enzymes in Humans and in Nonclinical Species.

- 1 many compounds, not one or two. This includes providing PK predictions when there are no
- 2 data, such as first in human studies. The methodologies involve physical chemistry, QSAR,
- 3 allometry, and empirical data on processes that govern clearance of drugs, mainly urinary or
- 4 fecal excretion and metabolism. Unbound fraction is a huge factor throughout the life stages (and
- 5 the physiological processes that control the unbound fraction). This methodology provides useful
- 6 tools, based on physiology, for extrapolations across age groups in humans. Efforts are ongoing
- 7 in the development of PBPK models for pregnancy, including the fetus and for the lactating
- 8 women, including the nursing infant. The value in exploring this modeling space is that human
- 9 data are collected in some cases, which is usually not the case for environmental chemicals.

However, it is not clear the extent to which databases are available for laboratory animals to implement this comprehensive approach for non-human species.

Justification for selection of the Verner et al. (2016) model

The draft MCLG documents state that the Verner *et al.* (2016) model was selected because “the Goeden *et al.* (2019...) model did not account for the decrease in concentration that occurs due to growth dilution which plays a substantial role in the PK of growing infants and children.” The lack of “growth dilution” in the Goeden *et al.* (2019) is not an adequate reason to disfavor this model, particularly since this factor is easy to add, and in any case (as noted by scientists who developed the model) represents a small correction. Additionally, the higher dose received by young children due to their higher drinking water consumption would tend to counteract the growth dilution effect. For example, Goeden *et al.* (2019; Figure 6) predicts that, from a certain concentration of PFOA or PFOS in drinking water, the serum PFOA or PFOS level in a 5-year-old child who was breastfed for 6 months is about 2.8-fold higher than at steady-state in adulthood. Serum levels at age 5 years would be even higher from breastfeeding for one year, as was assumed by EPA (p. 333, last paragraph of PFOA document).

The Panel questioned the choice of the Verner *et al.* (2016) model for several reasons. As discussed in the general comments on TK, there was a lack of availability of adequate information (model code, parameters, data, and performance) to fully evaluate the model (though the Panel does note the published model fit for PFOA appears to overpredict at 3 months and underpredict at 3 years). Additionally, if the model parameters were changed by EPA (see previous comment), then the adequacy of the model fit remains remaining the same would be questioned. Finally, and most importantly, the Verner *et al.* model assumes a constant oral dose, as opposed to a constant drinking water concentration, which may be more appropriate given the context of deriving an MCLG (discussed next).

Consideration of maternal and child drinking water ingestion rates

In EPA’s draft approach, it is unclear how a “dose to mothers & children that results in the same serum concentration at 5 years of age” can account for the different drinking water consumption rates in adults (mothers) and children. All the TK modeling used for deriving HEDs seems to be based on a constant dose rate in mg/kg-d. However, a constant dose rate will not equal a constant drinking water concentration due to age-dependent changes in drinking water consumption and lactational transfer. Therefore, if the goal is an equivalent “internal dose” POD based on AUC or C_{max} , then different values will result when using dose rate first and then converting to drinking water concentrations versus drinking water concentrations directly. Otherwise, the RfD will not be protective of all life-stages.

For instance, the draft MCLG documents discuss differences between the Verner *et al.* (2016) and Goeden *et al.* (2019) models for predicting serum PFAS levels in early life. However, the draft MCLG documents do not recognize that the two models have different purposes and provide different information. The Verner *et al.* (2016) model predicts infant and child serum PFOA or PFOS levels resulting from a constant daily PFOA or PFOS dose (ng/kg/day) to the mother and to the child after weaning. However, it is not clear how a RfD from the Verner *et al.*

(2016) model, which predicts serum PFOA or PFOS levels at age 5 years from a constant daily dose to the mother and the child, can be used to develop an MCLG than considers both exposure through breastfeeding and post-weaning and the changing drinking water consumption rates up to age 5.

By contrast, the Goeden *et al.* (2019) model considers both age-specific toxicokinetic factors and the changing drinking water intakes at different age periods. Although the Goeden *et al.* (2016) publication presents the application of the model to PFOA, the model can also be applied to other PFAS by using chemical-specific values for half-life, volume of distribution, and other chemical-specific factors. As reviewed in Post (2021), at least four states (MN, NH, MI, WA) have used this model to develop drinking water guidelines for PFOA, PFOS, PFNA, and/or PFHxS. The Goeden *et al.* (2019) model predicts the serum PFOA or PFOS levels at any age (including infancy, childhood, and adulthood) that result from maternal and child consumption of drinking water with a certain concentration (ng/L) of PFOA or PFOS. Specifically, it considers the maternal drinking water intake rate which impacts PFOA or PFOS levels in breastmilk, and the varying drinking water intake rates in children of different ages after weaning. The daily water intake (L/kg/day) in young children is much higher than in older individuals, and it varies during different childhood age periods up to age 5.

Recommendations

The Panel recommends the following with respect to the use and application of the TK model for endpoints observed in human neonates or children:

- Although the draft MCLG documents develop RfDs, not MCLGs, EPA should develop a RfD based on serum PFOA levels that can be used to develop a drinking water concentration (MCLG) that is protective for all life-stages.
- EPA should reconsider its choice of the Verner *et al.* (2016) model and consider whether the Goeden *et al.* (2019) model is more appropriate for use in development of the PFOA and PFOS RfDs and MCLGs. While the Verner *et al.* (2016) model predicts dosimetry from a constant daily dose, the Goeden *et al.* (2019) model considers age-specific toxicokinetic factors (e.g., volume of distribution) and exposure factors (milk and drinking water intake), in addition to appearing to have equal or better model fits as compared to the Verner *et al.* (2016) model. Thus, the Goeden *et al.* (2019) model appears more “fit for purpose” for deriving drinking water MCLGs.
- Additionally, in the Goeden *et al.* (2019) study on PFOA, the “internal dose” POD was further adjusted for inter-species and intra-species uncertainty/variability so that the “RfD” was expressed on a dose metric equivalent, which then could be converted using TK modeling to either an equivalent external dose or an equivalent water concentration, as appropriate. EPA should take this approach, as it would better account for life-stage-specific changes in ingestion rates at a fixed water concentration that would be the basis of an MCLG.

C. *The key chemical-specific parameters that describe the transfer of the chemical from the mother to the child during gestation and lactation are the maternal to fetal serum ratio and the ratio of maternal serum to milk PFOA/S concentration. These ratios were assumed to be constant during gestation and lactation, respectively. Another important parameter is the rate of milk ingestion, which is chemical-independent and varies throughout lactation. Please comment on the strengths and weaknesses of the choice of parameters for fetal to maternal partitioning and partitioning into breastmilk, as well as the choice for lactation rate. Please also comment on the choice to assume that fetal to maternal partitioning and partitioning to breastmilk did not vary in time. Please describe whether there are other methods you would recommend to account for these changes over time and across development.*

The Panel agreed that using constant ratios for maternal to fetal serum and maternal serum to milk was reasonable given the available data. Mechanistically, the movement of PFOA or PFOS from blood, across the mammary tissue into milk and the movement back into the blood supply, involves both diffusion and active transport, and thus are not at a steady state condition each time nursing occurs. The compartment empties and is replenished with fresh milk. For long-lived chemicals, the influence of this dynamic compartment on steady-state assumptions is assumed to be minimal when evaluated over days/months. The ratio (milk/plasma) would also vary to some degree with milk composition, and publications for estimating the milk/plasma ratio for drugs, including acidic drugs, are available. Thus, although the assumptions of a fixed ratio for the fetus and breast milk are not correct for the time-scale of each occurrence of nursing, they are probably adequate for the long-lived chemicals, PFOA and PFOS.

Recommendations

None.

Charge Question #4 - Animal Toxicokinetic Model

A. *After a review of the available toxicokinetic models for PFOA/S predictions in laboratory animals, EPA selected the Wambaugh et al. (2013) model because it was parametrized using all species of interest, demonstrated good agreement with training and test datasets, and used a single, biologically motivated, model structure across all species. Does the panel agree with selecting this model? If not, please describe the rationale for alternative recommended approaches for the calculation of the internal dose metrics in adult animals.*

The Panel agrees with the selection of the Wambaugh et al. 2013 model for the calculation of internal dose metrics. The rationale for the choice of the model is clearly described. If a decision is made to use RfDs based on animal studies in the final document, the animal model will become an important part of the basis for the MCLGs.

Wambaugh et al., 2013 is a comprehensive toxicokinetic model – using 22 studies that include *in vitro* and *in vivo* dosimetry data. The *in vivo* studies include datasets for both PFOA and PFOA from mouse, rat, and monkey. The *in vivo* studies include single and repeat dose studies. Most of

the rodent studies used are well powered with $n=5-25/\text{group}$. Three *in vitro* assays were also included in the analysis. The Panel did not suggest another model.

A general comment about development of PODs from animal studies is that serum/plasma data from the study itself (e.g., from at the end of the dosing period), rather than the serum levels predicted by modeling, could be used when appropriate serum or plasma PFOA/PFOS data are available. The Panel suggests that EPA consider the pros and cons of this approach, since use of the data from the study itself could reduce uncertainty. It should be noted that four of the eight states whose PFAS drinking water guidelines were reviewed by Post (2021, <https://setac.onlinelibrary.wiley.com/doi/full/10.1002/etc.4863>) used an approach based on serum PFAS levels measured at the end of dosing in the development of RfDs for PFOA and PFOS.

Additionally, the PFOA document (p. 330) states: “The purpose of the animal PBPK model is to make predictions of internal dose in lab animals used in toxicity studies or in humans. Therefore, to evaluate its predictive utility for risk assessment, a number of dose-metrics across life stages were selected for simulation in a mouse, rat, monkey, or human.” However, toxicokinetic parameters appear to be presented only for animal species but not humans in Section 4.3.1, and application of the model to humans is not discussed. As such, it should be clarified whether or how this model was used to simulate human exposures.

Recommendations

The Panel recommends that EPA consider using measured serum/plasma data when available rather than levels predicted by model.

B. The animal model parameters were obtained through a Bayesian inference parameterization which produced wide credible intervals for some parameter values, but relatively tight credible intervals for the predicted serum concentration. Does the panel agree with using the median values of the estimated animal parameter distributions for prediction of serum concentration and internal dose metrics?

The Panel agrees with the use of median values for deriving the HED. However, the uncertainty in both the human and animal models predictions should be characterized, particularly in terms of their impact on the HED calculation. This is particularly important if a probabilistic approach were to be used to calculate risk-specific doses instead of the traditional deterministic RfDs. When parameter values greatly exceed biological plausibility consider placing bounds on the parameters.

Recommendations

The panel recommends that EPA characterize the uncertainty associated with using median predictions.

C. *Based on visual inspection of model predictions to the calibration datasets, EPA utilized sex-independent parameters for PFOS. The male-specific parameters were used for all rat-specific PFOS predictions including predictions in pregnant and nursing dams and the female-specific parameters were used for all mouse-specific PFOS predictions because the parameter values obtained from fitting the female-specific rat data and male-specific mouse data were not consistent with the overall TK parameters for PFOS and produced poor fits to the training and test datasets. Does the panel agree with this approach and justification for this assumption for PFOS? If not, please describe other approaches that could be considered?*

The Panel agrees that there are insufficient studies for proper modeling. Consider re-calibration, versus selecting a sex. The Panel suggests that EPA plot the model predicted plasma concentrations versus the observed or measured plasma concentrations with a unity line and lines representing 0.5 and 2x around the unity line. This would provide an easy way to visualize the data and judge performance.

Recommendations

The panel recommends that EPA plot the model predicted plasma concentrations versus the observed or measured plasma concentrations to better visualize model performance.

D. *EPA assumed a one compartment model for the developing infant based on the lack of infant-specific toxicokinetic data from rats and mice. This model utilizes averages of half-life and volume of distribution from the literature coupled with physiologically relevant lactational parameters for pup nursing. Does the panel agree with the decision to use this model structure for infant animals? If not, please provide data on infant-specific changes during the animal lactational-period that could be used to account for toxicokinetic differences between the adult and infant rats and mice.*

The Panel agrees that a one compartment model for the developing pup or infant is adequate for deriving HEDs from experimental animal studies. Little is known regarding transfer of PFOA from placenta or PFAS to breastmilk with active transport. While this is likely and feasible, this has yet to be demonstrated. However, some consideration could be given to the fact that neonatal elimination of PFOA could likely be lower than adults because renal expression of transporters is low after birth. With regard to PFOA clearance in the neonate during lactation, the assumption is that clearance would be similar to that of adults at low doses. This might be the case, but it should be considered that in rats, renal Organic Anion Transporter (OAT1 and 3), and Mate1 mRNA expression are substantially lower in fetal kidney. Expression then increases during the neonatal period, with increases at weaning and adulthood (Xu *et al.*, 2017). The Panel suggests the use of covariates such as ontogeny of transporters that would greatly improve the model as information becomes available.

Recommendations

None.

E. Several parameters dictate the transfer of chemical from the mother to her pup. Does the panel agree with the selection of these parameters for the animal model? If not, please provide your justification and alternative parameters.

The Panel agrees with the lactation model (e.g., Loccisano *et al.*, 2013). The parameters that dictated transfer of chemical from mother to pup were Maternal Milk: Blood Partition Coefficient (P_{milk}), Fetus:Mother Concentration Ratio (R_{fm}), species-specific *in vivo* determined half-life ($t_{1/2}$), V_d for PFOA and the species-specific milk consumption rate during lactation. The parameters and assumptions used are up to date with regard to publications and our knowledge of PFOA transfer from mother to pup.

Recommendations

The Panel recommends the following two analyses to better justify its parameter choices.

1) Sensitivity analyses or uncertainty analyses would be helpful to better characterize the impact of uncertainty of these parameters.

2) Lactational transfer involves movement into the milk and from milk back into the mother's blood supply. You assume the movement is unidirectional into milk. EPA should evaluate the impact of not accounting movement from milk back into the blood supply.

F. For neonatal animals, EPA assumed no sex differences in clearance in neonatal animals based on the lack of identification of sex-dependent differences in PFOA/S toxicokinetics from the available data. Does the panel agree with this assumption? If not, please provide your justification and available data on sex differences in neonatal rats.

The Panel agrees that the data are inadequate to parametrize sex differences in neonatal animals using the computational methods employed by the EPA. There is little to no information to argue that sex-dependent differences in PFOA/S toxicokinetics exist in neonatal animals. There is minimal to no evidence to suggest sex-specific mechanisms are present that could affect toxicokinetics (i.e., renal expression of transporters, renal reabsorption). There are a few publications that examine sex-specific OAT expression in neonatal mouse and rat kidney (Buist *et al.*, 2004; Buist *et al.*, 2002). However, these studies have not evaluated the toxicokinetics of PFOA or PFOS in neonatal animals.

Recommendations

None.

Charge Question #5A- Epidemiological Study RfD Derivation
EPA evaluated potential confounding as part of their study quality evaluation of the epidemiological studies and selected only 'medium' and 'high' quality studies for POD derivation. Have the epidemiological studies that were selected for dose-response modeling

sufficiently addressed confounding? If not, are there key additional analyses that could be performed to further address the potential confounding of PFAS exposures in these studies?

The potential for confounding that results in a measured association deviating from the causal effect has been addressed to varying degrees in the epidemiologic studies. The magnitude of confounding may vary across study populations and with few exceptions, has not been found to be substantial once basic demographic and social factors are considered. Known determinants of biomarkers of PFAS in blood include age, sex, body mass index, pregnancy, and breastfeeding history (for women), diet, and some health behaviors. Depending on their relationship to health outcomes of interest, these would constitute potential confounders.

Beyond these familiar concerns, there are some particular issues that need to be considered in addressing epidemiologic studies of PFAS to be used for dose-response modeling. While not all can be readily addressed, the Panel encourages EPA to carefully consider these potential sources of confounding with an explanation of how they were addressed in the studies and whether it seems likely that residual confounding is present. Because of the susceptibility of any given study or set of studies on a particular health endpoint to be affected by confounding, consideration of multiple endpoints would be beneficial. Considering multiple studies of a variety of endpoints in different populations would provide convergent evidence that is more reliable than any one study or health endpoint in isolation.

In considering the adequacy of the control for confounding, the Panel noted several different issues, including:

Correlated exposure to other forms of PFAS

Individual forms of PFAS do not occur in isolation and there tends to be a correlation among them, i.e., those areas and individuals with elevated exposure to one of the chemicals often have elevated exposure to others within that class. While not necessarily acting as a conventional confounder (i.e., an independent cause of the outcome), this would distort the quantitative estimates for dose-response modeling. An effect attributed to a given change in one form of PFAS might in fact be in part a function of other forms of PFAS that are associated with it. If this were the case, then the actual potencies of PFOA or PFOS would be lower than implied by the studies. The Panel suggests that this possibility be addressed more explicitly.

Therefore, information on co-exposures to other PFAS should be presented for each of the epidemiological studies selected for POD derivation, noting whether the impact of other PFAS was evaluated, and, if so, the results of the evaluation. The BMDs for decreased antibody response to vaccines that was published by Budtz-Jorgensen and Grandjean (2018) were selected as PODs for the final PFOA and PFOS RfDs. Budtz-Jorgensen and Grandjean (2018) addressed co-exposure to PFOA and PFOS and stated that this was accounted for in the development of BMDs for PFOA and PFOS.

EPA also selected five human studies for development of PODs for decreased birthweight caused by PFOA, and four of these five studies were also used for development of PODs for

1 this effect for PFOS. Information on the potential impact of PFOA and other PFAS on
2 associations of PFOS with decreased birthweight, and *vice versa* for PFOA, in each of these
3 studies should be discussed in the draft MCLG documents. For example, Chu *et al.* (2020)
4 evaluated the impact of adjustment for PFOA and PFOS on the effect of another PFAS (i.e.,
5 Chlorinated Polyfluorinated Ether Sulfonate, CIPFESA), but it appears that the impact of PFOA
6 on PFOS and *vice versa* were not evaluated. Additionally, it appears that Sagiv *et al.* (2018) did
7 not evaluate potential confounding by co-exposure to other PFAS; the HAWC evaluation states
8 that "there is some minor concern over potential bias due to confounding by other PFAS." The
9 sensitivity analysis conducted by Starling *et al.* (2017) does not appear to support an association
10 for PFOS with birthweight after co-exposure to other PFAS is considered. This sensitivity
11 analysis is not mentioned in the HAWC evaluation of this study, and the Panel suggests that it
12 should be reviewed by EPA to determine if this study is appropriate for dose-response for the
13 effects of PFOS on birthweight. Finally, as noted in the HAWC file, Wikstrom *et al.* (2020) did
14 not consider confounding by co-exposure to other PFAS; the authors discuss this as a limitation
15 of their study.

16 Additionally, BMDs from the same human study (Dong *et al.*, 2019) were used as PODs for
17 increased serum cholesterol for both PFOA and PFOS. As stated in the HAWC evaluation for
18 Dong *et al.* (2019), there was "no discussion of potential confounding across PFAS."

19 *Role of shared physiology in affecting biomarkers*

20 Most epidemiologic studies of PFAS come from the general population and are largely
21 reflective of background variations in exposure. There are relatively few studies from
22 locations in which there are pronounced differences in water PFAS concentrations or
23 differing occupational exposures to workers. The general consistency of results of studies
24 from these different types of populations is a strength of the epidemiologic database for
25 PFOA and PFOS. However, variation in physiological factors that affect toxicokinetics (e.g.,
26 kidney function, plasma volume expansion) will affect biomarkers of exposure, and the
27 possibility that these physiological factors are also associated with health outcomes associated
28 with PFAS must be considered as well. For example, in studying of PFAS and kidney
29 disease, there is the potential for those who have impaired kidney function to have elevated
30 PFAS levels with the kidney malfunction causing the elevated PFAS levels rather than the
31 reverse (Watkins *et al.*, 2013), and similarly for outcomes like early menopause (Dhingra *et al.*, 2017).

33 The pitfalls of relying on exposure biomarkers, rather than exogenous exposures, for assessment
34 of exposure to environmental contaminants in general were carefully and thoughtfully examined
35 by Weisskopf and Webster (2017). The concern here is that measured PFAS levels in blood is
36 that they are determined not just by the exogenous exposure to PFAS in water, food, consumer
37 products, air, etc., but may also be considerably impacted by physiologic variation in uptake and
38 excretion. Studies generally do not measure exogenous exposure but rather variation in blood
39 PFAS levels across individuals with differing biomarker levels. To the extent that these blood
40 PFAS levels reflect physiologic differences, there is a risk of confounding that applies to
41 essentially all the studies considered.

1 The susceptibility to “physiologic confounding” is greatest when no environmental basis for
2 differing levels among study participants has been identified. Therefore, the relatively few
3 studies that are from populations with a wider range of exposures, such as those based on
4 populations with differing water contamination levels (e.g., in the mid-Ohio Valley (Steenland *et*
5 *al.*, 2009) and in Ronneby, Sweden (Andersson *et al.*, 2019), are less susceptible to this bias even
6 when they rely on biomarker levels rather than concentrations of PFAS in the water. There is
7 some possibility that individuals with higher concentrations of PFAS in drinking water also have
8 differences in PFAS exposure due to differing levels of PFAS in their diets or indoor
9 environments or water consumption, but these are likely to be subtle and modest influences, so
10 that there is likely to be less impact of associations with health outcomes due to physiologic
11 confounding in these studies.

12 Studies that use clinical biomarkers are more susceptible to this than studies of clinical disease.
13 When the product of the study is a correlation of blood levels of PFAS with blood levels of
14 antibody, cholesterol, or liver enzymes, the possibility that physiological factors may potentially
15 jointly affect PFAS and clinical outcome measures should be considered. Birthweight studies are
16 clearly susceptible since it is well-established that greater plasma volume expansion is associated
17 with greater birthweight (Salas *et al.*, 2006) and likely also associated with lower (diluted) PFAS
18 levels, which would create a spuriously elevated estimate of the quantitative impact of PFAS on
19 birthweight. A recent examination of the literature (Steenland *et al.*, 2018) provided indirect
20 support for this hypothesized bias, with studies that measured PFAS exposure later in pregnancy
21 when this would have the greatest effect showing the strongest association with reduced
22 birthweight. Additionally, Verner *et al.* (2015) concluded that some, but not all, of the decreased
23 in birth weight associated with maternal PFAS is accounted for by differences in maternal
24 glomerular filtration rate and that this effect may be greater in studies based on blood serum
25 PFAS measured later in pregnancy. A new study by Chang *et al.*, 2022 links PFOA (and PFNA)
26 with changes in both metabolic pathways and decreased fetal growth (small for gestational age)
27 in pregnant African American women (i.e., a human study), and attempted to examine
28 biomarkers less prone to reverse causation.

29 There is no direct way to overcome this uncertainty since too few studies have included
30 populations with clear exogenous sources that drive differences in PFAS levels. To the extent
31 that some studies are based on elevated environmental exposures as opposed to being based on
32 blood PFAS levels in the general population that are likely more impacted by physiologic
33 determinants, these would be preferred.

34 Additionally, studies of birthweight that measure PFAS before or early in pregnancy are more
35 informative than those that measure exposure later in pregnancy. If there are studies that provide
36 some temporal separation of the measures of PFAS and clinical biomarkers, those would be
37 preferred to those that measure them simultaneously. The draft PFOA document (p. 45) and the
38 PFOS document (p. 43) state that: "More confidence was placed in the epidemiologic studies [of
39 birthweight] that adjusted for glomerular filtration rate in their regression models or if they
40 limited this potential source of confounding by sampling PFAS levels earlier in pregnancy."
41 However, the consideration of these factors in regard to the specific studies of this effect selected
42 for POD development is not clearly discussed in the draft MCLG documents, and The Panel
43 recommends that this discussion be added.

Because susceptibility to confounding of the sort described above would likely differ across the endpoints of interest, this provides a strong justification to consider multiple endpoints rather than just one – that is, consider antibody response to vaccines, serum lipids, liver enzymes, and birthweight, not just one endpoint or one study from among an array of broadly similar ones. Several EPA IRIS assessments, including benzo(a)pyrene (U.S. EPA, 2017) and trichloroethylene (U.S. EPA, 2011) present RfDs for several different non-cancer effects. The final RfD that was selected was supported by generally similar RfDs for other non-cancer effects.

Role of SES and potential for residual confounding

Social factors often are associated with exposure to environmental contaminants and that possibility exists in some of the study populations. The assessment calls for thorough analysis for the presence of confounding and evaluating adjustments, specifically to determine whether residual confounding is present.

Potential residual confounding by socioeconomic status (SES) often contributes substantially to an overall low confidence rating for a study. “SES” is loosely used throughout the document and is a broad term with many contextual components. It would be helpful develop this further to consider which components of SES are suspected to confound associations of PFAS with particular health endpoints and if studies adjusted for potential confounding by *any* of these SES components *versus* none. Assuming that any and all components of SES will always confound associations with PFAS is not supportable. For instance, PFAS from drinking water does not necessarily correlate with lower SES, depending on the characteristics of the population served by different water sources. Using incomplete adjustment for SES as the primary basis for exclusion of studies from further consideration irrespective of the particular circumstances and sources of PFAS in each study is not warranted.

Most studies have been conducted among predominantly White populations. Levels of individual serum PFAS levels in the general population have been shown to vary by race/ethnicity, as demonstrated in analyses of data from multiple NHANES cycles (Calafat *et al.*, 2007). In general, PFAS levels in NHANES are lowest in Mexican Americans. Non-Hispanic White individuals have been consistently shown to have higher levels of PFOA compared to non-Hispanic Black individuals in NHANES, while for PFOS the levels are similar or somewhat lower among non-Hispanic Black compared to White individuals in NHANES. Additional analyses (Park *et al.*, 2019; Ding *et al.*, 2020) have also shown the importance of geography in evaluating racial disparities in PFAS exposure and serum concentrations, as well as differences in rates of decline of PFAS in blood by racial group. These studies also highlight PFAS serum level disparities between Asian populations and other racial/ethnic groups. Sensitivity analyses restricted to studies that included or separated diverse racial/ethnic groups would be worth considering; at the very least, potential confounding by race/ethnicity should be acknowledged particularly as some of the studied health endpoints may also exhibit racial/ethnic disparities.

Recommendations

The Panel has the following recommendations to improve the treatment of confounding:

- Consider multiple studies of a variety of endpoints in different populations so as to provide convergent evidence that is more reliable than any single study or health endpoint in isolation.
- Closely review sensitivity analyses and limitations noted by study authors, such as in Starling et al. (2017) and Wikstrom et al. (2020) as noted above, in determining if a study is appropriate for dose-response assessment.
- Do not use incomplete adjustment for SES as the primary basis for exclusion of a study without establishing that within the context of the study, SES is likely to be a confounder.
- Acknowledge potential confounding by race/ethnicity for health endpoints that may exhibit racial/ethnic disparities.

Charge Question #5B- Epidemiological Study RfD Derivation

Studies of developmental immune health outcomes (Grandjean et al., 2012 [HERO ID: 1248827]; Grandjean et al. 2017 [HERO ID: 3858518]; Grandjean et al., 2017 [HERO ID: 4239492]; and Budtz-Jorgensen and Grandjean, 2018[HERO ID: 5083631]) after PFOA/S exposure identified associations with very low doses of either PFOA or PFOS with developmental immune effects. The RfD for this outcome was selected as the critical effect because it was the lowest among the candidate RfDs for PFOA or PFOS and can result in severe illness. Does the panel agree with the selection of the critical study and critical effect for the derivation of chronic RfDs for PFOA and PFOS?

i. If so, please explain your justification.

ii. If not, please provide your rationale and detail an alternative critical study and/or critical effect you would select to support the derivation of chronic RfDs.

iii. Are any additional analyses or rationales needed to increase the confidence in the chronic RfDs for PFOA and PFOS?

General Comments

Overall, the Panel agreed with the selection of the critical study, Grandjean *et al.* (2012), and the critical effect, suppression of a vaccine response in children exposed during development, as appropriate for the derivation of chronic RfDs for PFOA and PFOS.

The critical effect is a reduction in antibodies produced in response to a vaccine in children exposed to higher levels of PFOA or PFOS compared to children with lower levels of exposure. Reduction in antibodies to a vaccine represents the failure of the immune system to respond to a specific challenge and is considered an adverse immunological outcome. The vaccine response is a functional response of the immune system to a specific challenge, in this case, an antigen (i.e., a non-self substance that stimulates a response from the immune system) delivered in the form of a vaccine. When a vaccine is delivered, cells of the immune system coordinate a response where the antigen is recognized, and cells of the adaptive immune system are ultimately stimulated to generate antibodies to the antigen. When the vaccine response is suppressed, it indicates that some part of the immune system is not performing at the level that it should. This form of

immunosuppression could indicate impacts on one or more parts of the innate and/or adaptive immune systems.

The critical effect of a suppressed antibody response to a vaccination is consistent with findings from additional epidemiological studies of different study populations across a range of vaccine types and also concurs with multiple studies in experimental animal models demonstrating that PFOA and PFOS are able to suppress the antigen-specific antibody response, the analogue to a vaccination in humans (NTP, 2016).

While the Panel agreed with the selection of the critical study and the critical effect, some expressed concerns with the process by which they were selected for the derivation of chronic RfDs for PFOA and PFOS. Among such process concerns were the need for a more thorough rationale for the selection of the critical study and critical effect and that additional candidate RfDs based on both human and animal studies should be presented. These concerns and accompanying recommendations are detailed in the remaining sections.

Critical Study

Several studies have been published investigating PFAS and vaccine responses in study participants from the Faroe Islands, the population upon which the critical study is based. These studies are somewhat homogeneous in that they shared not only a study population but were conducted by the same core group of researchers who designed and interpreted the studies in similar ways. The Panel recommends that supporting and contrasting data from study populations from the Faroe Islands and other locations be included in tabular form in, for example, Sections 3.3.4.1.1 of both documents where evidence of immunosuppression in human populations is presented. In addition to the epidemiological studies discussed in the draft document, the summary tables should also include the studies of decreased antibody response to vaccination that are not discussed in the draft MCLG documents, such as, earlier studies included in the 2016 HESD and more recent studies recommended by Panel members. Including such additional information would improve clarity and transparency for the selection of the critical study and would verify that the critical study and its study population are congruous with studies evaluating vaccine responses in other populations and not simply an outlier.

Additionally, while data from publications that studied participants from the Faroe Island (Grandjean *et al.*, 2012; 2017a, 2017b; Mogensen *et al.*, 2015) are summarized in the text and listed in Figures 61 (PFOA) and 62 (PFOS), it is challenging to understand the design of the study/studies from the information presented in the draft MCLG documents. These four publications provide evaluations of one, two, or both study cohorts that include subjects born in different time periods, with measurements of serum PFAS concentrations collected from maternal or child blood at different times/ages, and responses to vaccines in children measured at different ages. For clarity, a table that presents the designs of these Faroe Islands studies is needed in the draft MCLG documents.

The Panel was also concerned that the rationale for the choice of the critical study and dataset used for RfD derivation was not provided. One limitation of using Grandjean *et al.* (2012) and other epidemiological studies examining decreased antibody responses to vaccines associated

with PFAS for derivation of RfDs was that these studies lacked information about the dose-response for PFOA and PFOS individually. This issue was subsequently addressed in the development of BMDs and BMDLs from the Grandjean *et al.* (2012) data in the modeling done by Budtz-Jorgensen and Grandjean (2018). This modeling accounted for the impact of co-exposure to PFOA, PFOS, and other PFAS on the dose-response for each individual PFAS. While the Panel noted that the availability of BMDs and BMDLs that accounted for the impact of other PFAS may have been the primary reason for the selection of Grandjean *et al.* (2012) as the critical study rather than one of the other studies of decreased vaccine response, no explanation for this choice is provided in the draft MCLG documents.

The Panel also noted the lack of clarity concerning why data for age five serum PFAS and age seven antibody response was selected instead of maternal serum PFAS and age five antibody response, which also are included in Grandjean *et al.* (2012). The fact that the PODs are based on age five serum PFAS data and age seven antibody responses does not appear to be mentioned in the text or tables in Section 4.0, although this is mentioned in Appendix B.1.1 where the BMD modeling is discussed. However, even in Appendix B.1.1, it does not appear to be mentioned that Budtz-Jorgensen and Grandjean (2018) also evaluated data for maternal serum PFOA and PFOS and age five antibody responses.

It also appears that the critical study (Grandjean *et al.*, 2012) did not undergo the full systematic review evaluation for the domains included in the HAWC. Presumably, only those human studies that were not included in the 2016 HESD underwent a full systematic review. As data from Grandjean *et al.* (2012) were used as the basis for the final RfDs, it should be included in the systematic review of studies of decreased response to vaccines.

The Panel also noted that neither the California EPA nor the European Food Safety Agency (EFSA) selected Grandjean *et al.* (2012) as the basis for their toxicity factors (RfDs and Tolerable Weekly Intakes, respectively). The Panel recommends that EPA review the draft MCLG documents produced by California EPA (2021) and EFSA (2020) as they may contain useful information salient to reviewing Grandjean *et al.* (2012).

Recommendations

The Panel recommends that additional clarification and detail, including summary tables, be included to support the selection of the critical study and why other studies demonstrating suppressed vaccine responses were not selected as the critical study.

The Panel also recommends that results of the systematic review evaluation of the critical study also be included; if it was not systematically reviewed, the review needs to be performed.

The Panel recommends that the conclusions from other agencies about Grandjean *et al.* (2012) be reviewed and potentially included.

1 *Other Critical Effects*

2 The Panel appreciates that effects used as the basis for an RfD must be well established, adverse
3 or a precursor to an adverse effect, and relevant to humans if they are based on data from
4 experimental animal studies. Information presented in the draft MCLG documents demonstrates
5 that decreased antibody responses to vaccinations is a well-established effect of PFOA and
6 PFOS; this conclusion is further supported by earlier studies included in the 2016 HESDs and
7 more recent studies as noted earlier. The Panel also agrees that decreased antibody responses to
8 vaccinations are adverse effects, and that this effect is an appropriate critical effect for deriving
9 RfDs for PFOA and PFOS.

10 The Panel notes that the volume and quality of evidence for immunosuppression is similar to
11 other health endpoints that have been consistently reported in epidemiological studies for
12 populations exposed to PFOA and PFOS. The Panel is therefore concerned that the preference
13 for decreased antibody responses to vaccinations over increased serum cholesterol, changes in
14 liver enzymes, or decreased birth weight is not clearly explained. Although the POD at a
15 modeled human equivalent dose (POD_{HED}) for decreased antibody responses to vaccinations in
16 children is lower than the POD_{HED} values for other endpoints for both PFOA and PFOS and
17 likely will be protective of sensitive subpopulations, additional justification is needed to support
18 the selection of this endpoint as the critical effect. While it is appropriate to seek to derive the
19 RfD from the most sensitive endpoint, selecting the lowest value without consideration of other
20 factors is not justified.

21 Because the strength of evidence and clinical relevance are similar for all four health effects
22 mentioned above, the Panel recommends that candidate RfDs be developed for multiple human
23 health endpoints for which appropriate dose-response data are available. Additionally, it is not
24 clear why PODs based on experimental animal studies were not also used to derive candidate
25 RfDs. Transparency would be increased by first deriving candidate values from all eligible
26 human and animal studies (with adequate confidence, etc.), and then subsequently selecting
27 health effect-specific and overall RfDs. Summary tables, as recommended for studies of vaccine
28 responses, are also recommended for other endpoints considered for RfD derivation. Table 15 in
29 both documents summarizes the studies across these endpoints considered for derivation of
30 points of departure (POD) but these tables do not include important study details, such as serum
31 PFAS concentrations.

32 Notably, it does not appear that health effect-specific RfDs were developed, even for the existing
33 candidate PODs. Such health-effect specific RfDs are clearly needed to implement the draft
34 mixture framework being reviewed concurrently. The strengths and limitations for each
35 study/POD across endpoints, including risk of bias, concordance or disagreement between
36 human and animal studies, and dose-response uncertainties, should be described to justify the
37 selection of the dataset used to derive RfDs for each health effect. Thereafter, selection of the
38 overall RfD from among the health effect-specific RfDs should also discuss the strengths and
39 limitations, including strength of evidence for hazard and the uncertainties.

40 Additionally, the Panel noted that in the benefits from reduction of cardiovascular disease
41 document also being reviewed separately by this Panel, there is a meta-analysis for total

cholesterol; it would seem straightforward to apply the same methodology to derive the beta-coefficients (“re-expressed,” if necessary, in units of per ng/mL) for antibody responses to vaccines and other health effect-specific endpoints. Such a coefficient could then be used for deriving PODs. Further, expressing study results in terms of a “slope” (e.g., beta coefficient) that could be used to derive a risk-specific dose should be considered. These “slope” values would be more amenable to economic benefit-cost analysis than PODs or RfDs.

Recommendations

The Panel recommends that additional clarification and detail be included to support the selection of the critical effect and why this effect, beyond having the lowest POD_{HED}, is the most scientifically appropriate choice as well as being the most protective of public health.

The Panel recommends that candidate RfDs be developed for other health endpoints that have been consistently reported in epidemiology studies, as well as from the PODs for effects in experimental animal studies.

The Panel recommends that the final choice of the health-effect specific RfDs and the overall RfD consider the strength and limitations of the data upon which each is based. A meta-analysis approach also should be considered.

Additional analyses or rationales

Clinical relevance of the critical effect

Decreased antibody responses to vaccines is relevant to clinical health outcomes and likely to be predictive of risk of disease. The conclusion that suppression of vaccine responses is an adverse finding is widely accepted in the field of immunotoxicology. Immunosuppression indicated by reductions in antibodies to an antigenic challenge such as measured by vaccine responses in humans is quite different from immunosuppression that can be classified as an immunodeficiency from, for example, a genetic condition or a disease that severely depletes cells of the immune system. The form of immunosuppression that is represented by a reduction in the antibody response is a form that is similar to those who are at the extremes of age (young and old), those who have received organ transplants and are on immunosuppressive therapies, and those who are exposed to chronic stress and, mild to moderate immune suppression increases the risk of infections with pathogens commonly encountered in the general population (Selgrade, 2007). Additionally, when the antigen-specific antibody response is evaluated in experimental animal models, results are translatable across multiple species, including rodents and humans. Historical data associated with suppression of this response also indicates that it is highly predictive of immunotoxicity (Myers, 2018). Moreover, the immunosuppression indicated by the observed antibody decreases are not limited to those specific antigens (e.g., tetanus and diphtheria only), but rather are indicative of modulation of the general immune response.

Additionally, epidemiological evidence for increased risk of infectious disease associated with PFOA and PFOS exposure is discussed in sections 3.3.4.1.1 (evidence of immunosuppression in human populations) in both documents. This effect is not, however, included in the evidence

integration sections of the draft MCLG documents and no strength of evidence conclusions are provided. While evidence for an association of PFOA and PFOS with infectious disease is mixed, a 2019 review of the immuno-toxicological literature used to support an immune-based toxicity value for PFOS in drinking water concluded that studies available through 2018 “provide evidence for an association between general population levels of PFOS exposure and infectious disease, a clinically meaningful measure of health risk” (Pachkowski *et al.*, 2019). Several more recent publications that address PFAS exposure and infectious disease/risk of infectious disease also were not included in the draft MCLG documents (Bulka *et al.*, 2021, Dalsager *et al.*, 2021, and Timmermann *et al.*, 2020).

Recommendations

The Panel recommends that additional information on infectious disease outcomes be added to the evidence integration sections of both documents including strength of evidence conclusions.

The Panel recommends that additional clarification and detail be included to better frame the finding of antigen-specific antibody responses as a well-established adverse outcome, in and of itself, even in the absence of definitive evidence of increases in infectious disease/infectious disease risk.

Use of epidemiological data rather than experimental animal data as basis for RfD

The Panel supports the use of human epidemiological data as the basis for RfDs for PFOA and PFOS and notes that several human health outcomes, including decreased antibody responses to vaccinations in children, increased serum cholesterol, changes in liver enzymes, and decreased birthweight, have consistently been associated with PFOA and PFOS exposure, including within the range of general population exposures. However, an expanded explanation of the rationale for the preferential use of human studies for POD derivation is needed, especially because PODs from human data are much lower than if animal data were used. It is important that the rationale for this decision be thoroughly explained and supported because it is a major shift from the approach used in the 2016 HESD, which concluded that human data were precluded from use for PODs and RfDs. This explanation should include strengths and limitations of PODs based on studies from both human and experimental animal models.

Recommendation

The Panel recommends that a clear and thorough rationale for the decision to base the RfDs on human data rather than on experimental animal data only be provided.

Application of toxicokinetic model

Although toxicokinetic models are discussed in other Charge Questions, the Panel raised some concerns that are applicable to additional analyses or rationales needed to increase the confidence in the RfDs for PFOA and PFOS. With respect to the toxicokinetic model, candidate RfDs themselves (as opposed to just the PODs) could be derived based on internal dose units (e.g., serum ng/ml). Using this approach, exposures at a constant drinking water concentration could

be modeled with the toxicokinetic model and appropriate exposure factors, and the MCLG could be derived as the dose for which the internal dose POD is not exceeded for a specified exposure duration. The distinction between drinking water concentration and oral dose is especially important because several health outcomes under consideration occur as a result of exposures during particular windows, such as *in utero* and during childhood. Exposure factors differ substantially during these periods from those for adulthood, for example.

Recommendation

The Panel recommends that candidate RfDs be expressed in internal dose units; these could then be converted either to traditional continuous oral dose units or to drinking water concentration units (e.g., for supporting a MCLG), the latter of which would consider varying exposure factors depending on lifestages. Alternatively, additional clarification and justification is needed to explain why the approach to convert to a continuous oral dose is adequately protective across lifestages.

Duration(s) of exposure to which RfDs apply

The PFOA and PFOS RfDs are based on an effect that results from shorter-than-chronic exposure (serum PFOA or PFOS levels at age five). These RfDs are considered to be protective for chronic exposures, as well as shorter-term exposures, because they are more sensitive (i.e., have lower PODs) than RfDs for chronic effects. This differs from many other contaminants (e.g., GenX) for which shorter-term (e.g., subchronic) RfDs are higher than chronic RfDs (U.S. EPA, 2021).

Although EPA stated, at the SAB meeting on December 16, 2021, that the RfDs apply to shorter-than-chronic as well as chronic exposures, the draft MCLG documents state only that PFOA and PFOS RfDs are intended for chronic exposures. Their application to shorter exposure durations needs to be clearly stated. In situations of drinking water contamination with PFOA and/or PFOS, the duration of exposure to which these RfDs apply has practical implications for the timeframe (e.g., acute, short term, longer term) in which exposure to contaminated drinking water needs to be stopped when the MCL is exceeded.

Relevant to this issue, the Panel notes that the U.S. EPA (2009) Provisional Drinking Water Health Advisories for PFOA and PFOS were stated to apply to short-term exposures. Additionally, the U.S. EPA (2016) Lifetime Health Advisories for PFOA and PFOS, which are based on developmental effects, are stated to apply to both short-term (weeks to months) and lifetime (chronic) exposure.

Recommendation

The Panel recommends that the durations of exposure to which the RfDs apply be clearly stated, with explanatory text. This is critical in addressing situations of drinking water contamination with PFOA and/or PFOS. These durations (and corresponding lifestages) should also be considered when implementing the above recommendation to convert RfDs from serum levels to drinking water concentrations.

1 *Benchmark dose (BMD) model and BMD level*

2 The Panel noted that EPA did not independently replicate the BMD modeling conducted by
3 Budtz-Jorgensen and Grandjean (2018). It also appears that neither the details of the modeling
4 nor the modeling output are included in the Budtz-Jorgensen and Grandjean (2018) publication.
5 The Panel was informed at the December 16, 2016 meeting that EPA did not independently
6 replicate the Budtz-Jorgensen and Grandjean (2018) modeling and the authors provided a
7 supplemental document with details of the BMD modeling that was reviewed by the EPA Office
8 of Research and Development. However, the EPA's review of this BMD modeling is not
9 discussed in the draft MCLG documents. The Panel concluded that it is critical to include the
10 unpublished supplemental document provided to EPA by the authors of Budtz-Jorgensen and
11 Grandjean (2018), as well as the details and conclusions of EPA's review of this BMD modeling
12 in the final MCLG documents. While it seems unlikely that the supplemental document contains
13 confidential information such as personally identifiable information associated with study
14 participants, a version with the confidential information redacted could be provided if such
15 information is included in the document.

16 As noted earlier and relevant to the modeling conducted Budtz-Jorgensen and Grandjean (2018),
17 there was no rationale provided as to why EPA selected the BMDs for age five serum PFOA or
18 PFOS and age seven antibody responses to vaccinations, rather than maternal serum PFAS and
19 age five antibody responses to vaccinations as the basis for the RfDs. In the absence of a
20 rationale, the reason for this decision is unclear because the BMDs for maternal serum PFOA
21 and age five antibody responses to vaccinations for the piecewise model (the model that was
22 selected by EPA) are lower, and the ratios between the BMDs and the BMDLs are smaller, than
23 for the BMDLs for the piecewise model for age five serum PFOA at age five years and tetanus
24 antibody responses at age seven years that was selected by EPA. Further, the rationale in the first
25 paragraph of section B.1.1 of the draft MCLG documents for selection of the BMDLs for the
26 piecewise model instead of the linear model is unclear.

27 **Recommendations**

28 The Panel recommends that EPA provide supplemental data from the Budtz-Jorgensen and
29 Grandjean (2018) publication used for BMD modeling as well the conclusions of EPA's review
30 of the modeling in the publication, and additional rationale for the selection of specific BMDLs
31 from this publication. It is essential that details of the BMD modeling that forms the basis of the
32 PODs is transparently available for evaluation of the methods, approaches, and results.

33

34 **Charge Question #5C. Epidemiological Study RfD Derivation**

35 *The health outcomes identified in the critical studies were decreased antibody response,*
36 *specifically in serum anti-tetanus and anti-diphtheria, in children after vaccination (Grandjean*
37 *et al., 2012 [HERO ID: 1248827]; Grandjean et al. 2017 [HERO ID: 3858518]; Grandjean et*
38 *al., 2017 [HERO ID: 4239492]; and Budtz-Jorgensen and Grandjean, 2018 [HERO ID:*
39 *5083631]). This health outcome represents an increased susceptibility to a disease that can*
40 *cause very severe symptoms, including lethality. Furthermore, children who are*

1 *immunocompromised may mount a lower antibody response and in turn, be more susceptible to*
2 *contracting the disease, if exposed than healthy children. Because this health outcome has the*
3 *potential for severe illness and was assessed in children (i.e., EPA guidelines [US EPA, 1991]*
4 *support a 5% BMR for developmental effects), a benchmark response (BMR) of 5% was*
5 *selected for benchmark dose modeling. While some clinical findings are available, the clinical*
6 *relevance of a 5% decrease in antibody response is not clear. Given the need to protect*
7 *sensitive subpopulations (e.g., children, individuals with pre-existing conditions) and the*
8 *available clinical data (i.e., antibody response clinical level), does the SAB support the 5%*
9 *BMR selection for modeling to identify the POD? If not, please recommend the BMR level and a*
10 *scientific rationale for an alternative selection.*

11 The Panel generally supports the use of 5% BMR for decrease in antibody response, however,
12 stronger justification of the significance and relevance of the decreased antibody response in
13 comparison with other adverse outcomes, e.g., decreased birth weight, elevated serum ALT, will
14 strengthen the rationale.

15 The 5% BMR is supported by the developmental effect and potential severity of vaccine-
16 preventable illnesses (i.e., tetanus and diphtheria). The developing immune system (i.e., the
17 immune system of children) is generally regarded as more sensitive to exogenous perturbations
18 such as those from chemical stressors than the adult immune system. Therefore, changes
19 observed in the developing immune system reflect developmental effects. The immune response
20 to childhood vaccines may be “an excellent indicator for developmental immunotoxicity when
21 conducted under appropriate conditions” (Luster *et al.*, 2005). Responses to childhood vaccines
22 are thought to be sensitive enough to detect changes in populations with moderate degrees of
23 immunosuppression, such as those exposed to an immunotoxic agent (Luster *et al.*, 2005).

24 Hessel *et al.* (2015) evaluated data from experimental animal studies of nine different chemicals
25 known or suspected to be immunotoxicants and that had been evaluated for immunotoxicity
26 across a range of tests, including guideline tests for regulatory toxicology testing. With respect to
27 the antigen-specific antibody response, which when performed in experimental animals is
28 analogous to the vaccine response in humans, Hessel *et al.* concluded that any compound-related
29 effect is indicative of adversity. Therefore, a BMR of 5% would likely be small enough to be
30 protective against shifts in the vaccine response that are considered to be small, mild, or
31 moderate, but that are still differentiated in some way from no or low exposure groups.

32 However, the degree to which mild to moderate immunosuppression from exposure to chemical
33 agents produces measurable clinical outcomes is challenging to determine (DeWitt *et al.*, 2017).
34 Evidence from specific populations experiencing mild to moderate immunosuppression indicates
35 that the risk of infections with pathogens commonly encountered in the general population is real
36 (Selgrade, 2007). Reductions in antibody titers to a specific vaccine below a level that is
37 considered protective does increase the risk of susceptibility to the disease against which the
38 vaccine was intended (McComb, 1964; MacLennan *et al.*, 1965; WHO, 2017; WHO, 2018).
39 Additionally, “a compromised immune system should be considered more prone to escape
40 homeostasis, enhancing risk for disease development” (Hessel *et al.*, 2015). The clinically
41 significant decrease in tetanus and diphtheria antibody concentrations is generally considered to
42 have antibody concentrations below 0.1 IU/mL. Considering the large number of people (30-

40%) having antibody concentrations close to 0.1 IU/mL, a further 5% decrease in antibody concentrations could be problematic for disease protection. More susceptible individuals may be more likely to be affected by the infection. The benchmark response of 5% is conservative for people at the lower end of antibody response.

EPA will need to reconcile the varying research studies that have been presented about how a reduced antibody response to toxoids will affect health protection against infection. While the research from the Faroe Islands is the basis that EPA used for BMD analysis, the Timmermann *et al.* (2022) study in Greenland needs to be considered. Additionally, studies of measles antibody response in Guinea-Bissau (Timmermann *et al.*, 2020) and Hepatitis A antibody response in the Faroe Islands (Shih *et al.*, 2021) need to be considered for transparency and consideration of other relevant health endpoints.

While decreased antibody response is an important developmental outcome, decreased birthweight needs to be considered as an alternate development outcome. The proposed RfD is based on studies of decreased serum antibody concentrations that are of medium quality. For PFOA, the candidate RfDs developed from high-quality studies of decreased birthweight are 3, 60, 70, 70, and 300 times larger than the candidate RfD based on serum antibody concentration (ranked by magnitude). For PFAS, the difference is smaller; the candidate RfDs for decreased birthweight are 20, 20, 20, and 100 times larger. The choice between a smaller RfD from lower-quality studies (all from the same research group and study population) and a larger RfD from higher-quality studies (from multiple groups and populations) requires a strong rationale and perhaps discussion of how much difference it makes to the result.

Recommendations

The Panel recommends that EPA provide a stronger and more transparent justification for the BMRs for not only decreased antibody response, but also other endpoints for which BMDs were developed. Ideally, BMR levels were correspond to a similar level of adversity or risk across endpoints.

Charge Question: #5D. Uncertainty Factors

EPA has evaluated and applied where appropriate uncertainty factors to account for intraspecies variability (UFH), interspecies differences (UFA), database limitations (UFD), duration (UFS), and LOAEL-to-NOAEL extrapolation (UFL) for PFOA and PFOS.

i. Has uncertainty been adequately accounted for in the derivation of the RfDs? Please describe and provide suggestions, if needed.

ii. Does the provided scientific rationale support the application of the selected uncertainty factors? Please explain.

The Panel noted that in section 4.1.5 of the PFOA (pp. 337 – 339) and PFOS (pp. 308 – 310) documents, EPA applied a value of 1 for interspecies (UFA), subchronic-to-chronic (UFS), LOAEL-to-NOAEL (UFL), and database (UFD) uncertainty factors. Justification for these uncertainty factors can be found in Table 22 of both documents (PFOA: pp. 337 – 8; PFOS:

p.308). Briefly summarized, a UF_A of 1 was selected because the $RfDs$ were developed with human data. A UF_S of 1 was selected because the critical effects (decreased antibody response to tetanus or diphtheria vaccine from exposure at age 5) were the result of a shorter-than-chronic exposure that is more sensitive than the chronic effects of PFOA and PFOS. A UF_L of 1 was selected because the $RfDs$ were based on a BMDL. Finally, a UF_D of 1 was selected because the database includes numerous medium- and high-quality studies and a more sensitive endpoint than the critical effect is not expected. For both PFOA and PFOS, EPA applied a default value of 10 for the intraspecies (UF_H) uncertainty factor to account for variability within human populations based on intrinsic and extrinsic factors that can influence response.

The Panel generally found these values to be adequate and supported by the scientific rationale provided by the Agency. The values were found to be appropriate and sufficiently protective, with rationale that was clearly described in the draft MCLG documents.

While the Panel recognizes that the EPA has time and resource constraints, they recommend that the Agency consider the adoption of a probabilistic framework (including UF distributions, rather than fixed values) to calculate risk-specific doses as a replacement for traditional $RfDs$, in line with the recommendations of the NASEM (2009) Science and Decisions report. A recent publication Chiu *et al.* (2018) demonstrated broad application of this approach using experimental animal studies across many chemicals and endpoints and included EPA authors from EPA's National Center for Environmental Economics. This probabilistic framework not only includes default UF distributions based on reviews of the literature, but also enables derivation of dose-response functions (or risk-specific doses) that can be used for benefit-cost analysis. EPA should consider whether applying this approach here would be useful for MCLG derivation and the regulatory impact assessments that will be needed to set MCLs.

Another suggestion is that EPA consider the appropriateness of using an additional uncertainty factor (either as a justification for an increase in an existing uncertainty factor, e.g., UF_H or UF_D , or as an independent UF) that accounts for the effects of simultaneous co-exposures to PFOA/PFOS and complex mixtures of chemical and non-chemical stressors. As the agency acknowledges, PFAS are known to occur in mixtures, with thousands of chemicals in the class. Although a UF_{TOT} of 10 could represent a conservative approach for PFOA and PFOS, it may not be fully protective of highly exposed and susceptible populations exposed to PFAS in drinking water.

One potential framework to explore is the Mixture Assessment Factor (MAF). The use of MAFs is discussed in Kortenkamp and Faust (2018) and is currently being explored by the European Commission (2020) for the assessment and management of chemical mixtures. A potential starting place for incorporating an MAF would be to create an UF_M that accounts for mixtures, with a default value equal to 10. The default value assumes that only a small number of chemicals contribute to a particular effect. The value of the UF_M could be increased or decreased based upon the number of chemicals (and to the extent possible, non-chemical stressors) expected to co-occur with the chemical being evaluated. Given that PFAS often co-occur, PFOA and PFOS are ideal chemicals for which to utilize a mixture-associated uncertainty factor with the default value of 10.

However, the Panel did not reach consensus on advising this approach for the current assessments because of divergent views on the appropriate methods of accounting for the effects of mixtures on chemical toxicity. Noting that the reference dose is specific to the intrinsic behavior of the chemicals being evaluated, in the absence of other chemicals that cause the same effect or inhibit its effect, there was concern about using this approach. Further stating that an uncertainty factor accounting for the effects of other chemicals on the intrinsic behavior of the chemical being studied was deemed inappropriate. The approaches discussed in the draft EPA Mixtures Framework document – in which co-occurring PFAS that cause the same toxic effect are considered – were seen as more appropriate for accounting for toxicity of mixtures of PFAS. In these approaches, a toxicity factor or relative potency is developed for each individual PFAS, and their additive effects are considered.

Others noted that uncertainty factors should have a clear conceptual basis before being applied. There were concerns that identifying an agreed-upon framework for developing a mixture uncertainty factor, should one exist, was beyond the scope of the charge. Furthermore, it could be argued that the concerns about mixtures are already covered by the UF_H , since human variability incorporates not just genetic variability, but variability in background and co-exposures, nutrition, etc. (Zeise *et al.* 2013).

In addition to considerations associated with the use of uncertainty factors, the Panel also recommends that as the Agency moves forward, it should consider cumulative risk, not just from multiplicity of chemical exposures, but from other environmental factors that enhance susceptibility – including co-morbidities and/or co-exposure to social disparities (including racial, economic, and power disparities). Of particular concern are populations that may have increased susceptibility across multiple chemical, biological, and social domains (Pullen Fedinick *et al.*, 2021). While cumulative approaches may be difficult to apply for the assessments of PFOA and PFOS for MCLG derivation, development of approaches for combined risks to health from multiple stressors is an important area of risk assessment that warrants further consideration.

Recommendations

The Panel recommends that EPA consider adoption of a probabilistic framework to calculate risk-specific doses, in particular as to whether applying this approach would be useful for MCLG derivation and/or regulatory impact assessments needed to set MCLs.

The Panel did not reach consensus on methods for accounting for effects of mixtures due to PFOA and PFOS usually occurring with other PFAS, but recommends that EPA evaluate the potential applicability of different approaches and their implications for setting MCLGs.

Charge Question #6 - Relative Source Contribution

EPA applies a Relative Source Contribution (RSC) when calculating the MCLG to provide a margin of safety that an individual's total exposure from a contaminant does not exceed the RfD. The RSC is the portion of an exposure for an individual in the general U.S. population

estimated to equal the RfD that is attributed to drinking water; the remainder of the exposure equal to the RfD is allocated to other potential sources. Based on the physical properties, detected levels, and available exposure information, there are significant potential sources other than drinking water ingestion for PFOA and PFOS; however, information is not available to quantitatively characterize exposure from these different sources. EPA followed Agency guidance on how to derive an RSC (U.S. EPA, 2000; available online at: <https://www.epa.gov/sites/default/files/2018-10/documents/methodology-wqc-protection-hh-2000.pdf>) and recommends an RSC of 20 percent (0.20) for PFOA and PFOS. This RSC is the same as what was used in the 2016 HAs for PFOA and PFOS.

- i. Are you aware of additional relevant exposure data that EPA should consider in developing the RSCs for PFOA and PFOS? If so, please provide citations.*
- ii. Please provide comment on whether the recommended RSC of 20 percent (0.20) for PFOA and PFOS is adequately supported and clearly described.*

The Panel concluded that the recommended Relative Source Contribution (RSC) of 20% in the draft MCLG documents is appropriate for the PFOA and PFOS Reference Doses (RfD) development. A 20% RSC is the default value specified in the U.S. EPA (2000) guidance for deriving ambient water quality criteria for the protection of human health, which is cited as the basis for the RSC selection in the draft MCLG documents. The U.S. EPA (2000) guidance recommends an RSC range with a minimum ("floor") of 20% and a maximum ("ceiling") of 80%. A lower RSC results in a more stringent and protective, water concentration, as represented by the MCLG. Hence the default value of 20%, referred to as the "floor" value in the U.S. EPA (2000) guidance, is the most stringent and public health protective choice.

While the Panel supports the selection of an RSC of 20%, the Panel recommends that EPA revise certain aspects of the RSC sections in the draft MCLG documents to better describe and explain the rationale for arriving at an RSC of 20%. This will also help ensure that the selection of the 20% RSC is consistent with the approach provided in the U.S. EPA (2000).

The most important concept that the Panel would like to reiterate is that the RSC represents the portion of the RfD that is allocated to drinking water. The intent of applying an RSC is that the total exposure from all sources at the MCLG will not exceed the RfD. The RSC is therefore dependent on the portion (percentage) of the RfD that is known or assumed to come from non-drinking water sources (e.g., food, consumer products, air, dust). Therefore, actual PFOA/PFOS exposures from drinking water, including the percentage of total PFOA or PFOS exposure that comes from drinking water and the concentrations of PFOA or PFOS in drinking water, are not relevant to RSC selection.

However, the draft MCLG documents describe the RSC both in terms of the portion of the RfD that is attributable to drinking water and in terms of PFOA/PFOS intake from water relative to total intake, where the latter is not relevant to the RfD. The RSC determination should not be based on the relative PFOA or PFOS water to non-water exposures, without the context of the RfD. In other words, the denominator in the RSC calculation, is the RfD, not the total PFOA or PFOS exposure.

The Panel therefore recommends that EPA revise the RSC section in the draft MCLG documents to promote a common understanding of the concepts and methodologies involved in the RSC determination as per the 2000 EPA Guidance, and that the document should clearly outline why the 20% value is the most appropriate one for the proposed PFOA and PFOS RfDs.

Specifically, statements in the draft document that suggest that the percent of exposure from drinking water is relevant to RSC selection should be removed. For example, Section 5.1.4 of the draft PFOA document and Section 5.4 of the draft MCLG documents discuss several studies (Hu *et al.*, 2019; East *et al.*, 2021; Gebbink *et al.*, 2015; Jogsten *et al.*, 2012) that estimate that the percentage of total exposure to PFOA and/or PFOS that comes from drinking water is less than 20%, and further state that these “estimates support a 20% RSC for drinking water.” Another example is the statement in the draft MCLG documents that “for disproportionately affected subpopulations, such as the occupationally exposed or site-impacted (e.g., by a particular source or industry) where there may be higher average PFAS concentrations in drinking water, it may be appropriate to apply an RSC greater than 20 percent if there is sufficient information to quantitatively characterize sources other than drinking water.”

Additionally, because the RSC is based on the portion of the RfD that comes from non-drinking water sources, the choice of the RSC depends on the numerical value of the RfD. The RSC will decrease as the RfD decreases since the non-drinking water exposures represent a higher proportion of a lower RfD. Because the RfDs used in the 2016 EPA Health Advisories and in state drinking water guidelines are several orders of magnitude higher than the RfDs presented in the draft MCLG documents, the RSCs used in the 2016 HESD and by states (discussed on p. 347-348 of the PFOA document) are not relevant to the selection of the RSC in the current draft MCLG documents.

These revisions and clarifications will also help address questions related to the relevance to RSC selection of factors such as protection of disproportionately affected subpopulations and sensitive subpopulations such as infants and children, PFOA and PFOS concentrations in drinking water and relative exposures from drinking water, and serum concentrations from biomonitoring data. These issues are discussed in the RSC section of the draft MCLG documents and have therefore been brought up by SAB panelists and in public comments. The Panel recommends that EPA should therefore clarify what, if any, roles these factors play in the selection of an RSC of 20% as recommended in U.S. EPA (2000) guidance, or if they are being referenced as supplementary information and are more directly accounted for elsewhere in the MCLG development.

Although the rationale for an RSC of 20% in the draft document needs clarification, available data on non-drinking water exposures to PFOA and PFOS clearly support the choice of an RSC of 20% for MCLGs based on the RfDs presented in the draft document. Estimates of daily dietary exposure to PFOA in studies from North America and Europe reviewed by DWQI (2017) range from 0.16 ng/kg/day to 6.2 ng/kg/day, and the lower bound estimate of median daily dietary exposure for adults from EFSA (2020) is 0.18 ng/kg/day, with higher values for younger age groups. The RfD of 1.5×10^{-9} mg/kg/day (0.0015 ng/kg/day) in the draft PFOA document is two orders of magnitude below the lowest of these dietary estimates. Similarly, PFOS RfD of 7.9×10^{-9} mg/kg/day (0.0079 ng/kg/day) is almost two orders of magnitude lower than the EFSA

(2020) lower bound estimate of median daily dietary exposure to PFOS in adults of 0.58 ng/kg/day. Additionally, there are non-drinking water exposures from other sources such as consumer products and house dust. Therefore, exposures from non-drinking water sources far exceed the RfD, indicating the choice of the default RSC of 20% is appropriate.

An RSC of 20% is also supported by data on serum PFOA and PFOS levels from the U.S. general population. The serum PFAS levels associated with the RfDs can be determined by applying an Uncertainty Factor of 10 to the $POD_{Internal\ Dose}$ (e.g., human serum levels at the PODs) of 1.7×10^{-4} mg/L (0.17 ng/ml) from Table 21 of the draft PFOA document and 5.4×10^{-4} mg/L (0.54 ng/ml) from Table 21 of the draft PFOS document. The serum levels associated with the RfDs, 0.017 ng/ml for PFOA and 0.054 ng/ml for PFOS, are far below even the lower percentiles for serum PFOA and PFOS in the U.S. general population in the most recent (2017-18) NHANES data. In the 2017-18 NHANES, the 5th percentile serum PFOA level, presumably representing those with little or no drinking water exposure, is 0.48 ng/ml, which is 28-fold higher than the serum level associated with the RfD of 0.017 ng/ml. Similarly, the 5th percentile serum PFOS level, also presumably representing those with little or no drinking water exposure, is 1.08 ng/ml, which is 20-fold higher than the serum level associated with the RfD, 0.054 ng/ml. Therefore, serum PFOA and PFOS levels in the lowest 5th percentile of the general population (presumably without exposure to contaminated drinking water) far exceed 100% of the RfD, supporting the default RSC of 20%.

Recommendations

While the Panel supports the selection of an RSC of 20%, the Panel recommends that EPA revise certain aspects of the RSC sections in the draft MCLG documents to better describe and explain the rationale for arriving at an RSC of 20%. This will also help ensure that the selection of the 20% RSC is consistent with the approach provided in the U.S. EPA (2000). Specifically,

- Statements in the draft MCLG documents that suggest that the percent of exposure from drinking water is relevant to RSC selection should be removed.
- The relationship between the selection of the RSC and the numerical value of the RfD should be made clear.
- EPA should consider using available data on non-drinking water exposures to PFOA and PFOS to clearly support the choice of an RSC of 20% for MCLGs based on the RfDs presented in the draft document.
- Similarly, EPA should also explain how an RSC of 20% is supported by data on serum PFOA and PFOS levels from the U.S. general population.
- EPA should clarify the relevance of the RSC selection, or more generally to the MCLG development, to factors such as the protection of disproportionately affected subpopulations and sensitive subpopulations, including infants and children.

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SECTION II- Mixtures approaches

EPA's Draft Framework for Estimating Noncancer Health Risks Associated with Mixtures of Per- and Polyfluoroalkyl Substances (PFAS)

Charge Question #1- Dose Additivity Assumption

The component-based mixtures approaches presented in the framework are based on dose addition. Traditionally, an assumption of dose addition for a mixture is based on components sharing a common mode of action (MOA) for a given health effect. However, EPA's supplementary guidance (EPA, 2000) states: "The common mode-of-action (MOA) assumption can be met using a surrogate of toxicological similarity, but for specific conditions (endpoint, route, duration)." This suggests that although the common MOA metric for application of dose addition is optimal, there is flexibility in the level of biological organization at which "similarity" can be determined among mixture components. As an emerging chemical class, MOA data is limited or not available for many PFAS. For purposes of a component-based evaluation of mixtures additivity for PFAS, EPA assumes similarity at the level of toxicity endpoint/health effect rather than MOA.

- A. Please comment on the appropriateness of this approach (dose additivity based on common endpoint of toxicity or health effect) for a component-based mixture valuation of PFAS under an assumption of dose additivity.*
- B. If common toxicity endpoint/health effect is not considered an optimal similarity domain for those PFAS with limited or no available MOA-type data, please provide specific alternative methodologies for integrating such chemicals into a component-based mixture evaluation(s).*

The per- and polyfluoroalkyl substances (PFAS) chemical class includes thousands of unique chemicals that exist in environmental media often as PFAS mixtures. Toxicological studies to inform human health risk assessment are largely lacking, both for individual and PFAS mixtures. Further, mode of action (MOA) data is limited, if not completely lacking, for the majority of PFAS. EPA's draft framework proposes a component-based approach to estimating the probability or magnitude of adverse non-cancer health outcomes for PFAS mixtures. EPA has developed this framework based on the assumption of dose additivity for the component-based approaches used to assess the risks of PFAS mixtures (i.e., HI, TOSHI, RPF, Mixture BMD) presented in the draft framework. The framework proposes that dose additivity can be applied for cases in which the given adverse outcome (i.e., toxicity endpoint/adverse outcome/effect) associated with exposure to 2 or more PFAS chemicals is similar.

Assumption of dose-additivity

The draft mixtures document presents information supporting the assumption of dose additivity for chemical mixtures in general, including mixtures of PFAS. Several examples are discussed in the draft mixtures document (e.g., dioxin-like chemicals, organophosphate chemicals) that alter shared pathways which typically produce *at least* dose additive responses. The information

1 included in the draft framework supports the conclusion that toxicological interactions of
2 chemical mixtures are frequently additive or close to additive. It also supports the conclusion that
3 dose additivity is a public health protective assumption that typically does not underestimate the
4 toxicity of a mixture.

5
6 The SAB Panel agrees with use of the default assumption of dose additivity when evaluating
7 PFAS mixtures that have similar effects and concludes that this assumption is health protective.
8 It is noted that the assumption of dose additivity can provide an estimate of composite effects
9 when individual PFAS are below their NOAELs. However, it is recommended that, when data
10 clearly indicate interactions other than dose additivity, the approach indicated by the data should
11 be used.

12
13 It is further noted that the physical-chemical, toxicological, and toxicokinetic properties of PFAS
14 are different than for other classes of chemicals that have been studied as mixtures, and that there
15 are still many unanswered questions about their interactions in mixtures. While the assumption
16 of dose additivity may be reasonable at low concentrations, factors such as competition for
17 transport may result in non-additive interactions at higher concentrations. Information on the
18 doses at which such transitions may occur is needed. The Panel recommends that EPA reevaluate
19 the default assumption of dose additivity as additional data become available.

20
21 As discussed in the draft EPA mixtures document, a recent EPA Office of Research and
22 Development (ORD) study of PFOA and PFOS (Conley *et al.* - Appendix A of draft EPA
23 mixtures document) indicates dose additivity for developmental toxicity of these two PFAS in
24 rats. Other studies that indicating a common MOA and dose additivity for PFAS are also
25 reviewed in the draft framework. For example, the draft EPA mixtures document discusses that
26 Wolf *et al.* (2014) reported additivity for PPAR- α activation in binary mixtures of PFOA and
27 four other PFAS in cultured cells transfected with the mouse or human PPAR- α receptor. While
28 the dose additivity assumption is recommended for the reasons discussed above, the Panel
29 suggests that the discussion of studies of toxicological interactions in PFAS mixtures in the EPA
30 mixtures document be expanded to also include studies that do not indicate dose additivity and/or
31 a common MOA for PFAS. Some of these studies are summarized below. Acknowledging and
32 including this information will increase transparency and characterization of the uncertainties
33 associated with the assumption of dose additivity.

34
35 For example, a recent paper not cited in the draft EPA mixtures document, Marques *et al.* (2021),
36 indicates that toxicological interactions of a mixture of PFOA, PFOS, and PFHxS in mice can be
37 additive, synergistic, or antagonistic for specific hepatic and metabolic effects after perinatal
38 exposure. Surprisingly, this appears to be the first mammalian study of defined mixtures of
39 PFAS to be published in a peer reviewed journal. As stated by Marques *et al.* (2021): "The
40 PFAS mixture had very distinct effects when compared to single compound treatment. With
41 regard to liver weights and liver to body weight ratios increases, the PFAS mixture data were
42 analogous to the effects seen with PFOA treatment. However, unlike PFOA, the serum ALT
43 level, did not increase in the PFAS mixture. In the case of liver lipids, only the PFAS mixture in
44 combination with HFD [high fat diet] feeding decreased total cholesterol in the pups and
45 increased total lipid in the pups. However, liver triglycerides were increased with all three single

PFAS treatments with the SD [standard diet], and in treatment with the PFAS mixture with SD, there was no change compared to control... These results suggest that there are multiple pathways in which PFAS could add, synergize, or antagonize specific effects, and warrants further investigation of dose response data with model predictions of additivity." These results also suggest that co-exposure to other PFAS may impact the toxicokinetics of individual PFAS, as follows: "PFOS levels in pup and dam serum were lower in the PFAS mixture compared to PFOS treatment alone." The Panel suggests that discussion of this paper be added to the EPA mixtures document.

Another recent study, Nielsen *et al.* (2021), that was not included in the draft EPA mixtures document did not find dose additivity for activation of PPAR- α by PFAS mixtures in cultured cells transfected with a full length human PPAR- α construct. The Panel suggests that discussion of Nielsen *et al.* (2021) be included in the final EPA mixtures document. Nielsen *et al.* (2021) found that the potency (EC50) for PPAR- α activation varied among the seven PFAS tested. They also reported that the efficacy (maximal PPAR- α activation compared to positive control) was lower for perfluorosulfonic acids (PFSAs) than for perfluorocarboxylic acids (PFCAs), and that a general concentration addition (GCA) model that considers differences in both potency and efficacy among PFAS predicts the PFAS interactions better than a RPF approach that considers only differences in potency. They further conclude that an effect summation model can also likely predict the interactions at low concentrations.

Additional studies that report non-additive interactions of PFAS include, Kjeldsen and Bonefeld-Jorgensen (2013) who studied PFAS activation of the estrogen and androgen receptor in a cultured cell line transfected with these receptors; Ojo *et al.* (2020) who studied effects of binary and ternary PFAS mixtures on cell viability of a human liver cell line, HepG2; Ding *et al.* (2013) who studied interactions of PFOA and PFOS in zebrafish; and Menger *et al.* (2020) who studied behavioral effects in zebrafish of nine PFAS individually and a mixtures of equal concentrations of all nine PFAS.

Assumption of similarity of toxicity endpoint rather than common MOA in mixtures evaluation

The Panel agreed with use of a similar toxicity endpoint/health effect instead of a common MOA as a default approach for evaluating mixtures of PFAS. This approach makes sense because multiple physiological systems and multiple MOAs can contribute to a common health outcome. Human function is based on an integrated system of systems and not on single molecular changes as the sole drivers of any health outcome. The Panel concluded that rather than the common MOA, as presented in the EPA draft mixtures document, common physiological outcomes should be the defining position. Consider a health outcome such as elevated blood pressure (not one for PFAS or PFOS but just a general example). It is known that there are many different physiological systems that contribute to regulation of blood pressure beyond the renin-angiotensin system (Joyner and Limberg, 2014).

Furthermore, many PFAS, including the four used in the examples in the draft EPA mixtures document and others, elicit effects on multiple biological pathways elicit effects on multiple

biological pathways that have common adverse outcomes in several biological systems (e.g., hepatic, thyroid, lipid synthesis and metabolism, developmental and immune toxicities). For clarity, the Panel recommends that the difference between a MOA and a health outcome be defined in the framework. Additionally, when data clearly indicate that an approach based on common toxicity rather than common MOA is not supportable, the approach indicated by the data should be used.

The Panel notes that the U.S. EPA (2000) mixtures risk assessment guidance states: “The common mode of action (MOA) assumption can be met using a surrogate of toxicological similarity, but for specific conditions (endpoint, route, duration).” However, it is not completely clear how “duration” is incorporated into the approaches based on similar toxic endpoint that are proposed in the draft EPA mixtures document, and this should be clarified.

Finally, although there is little or no MOA data for many PFAS, information from *in vivo* studies indicates that the mode(s) of action for several key toxicological effects differ among several well-studied PFAS. That being said, dose additivity for a common toxicological effect can still apply even if the MOA for the effect differs among chemicals in a mixture. For completeness, it is the Panel suggests that a summary of information indicating that different MOAs for PFAS be included in the framework. For example, PFOA, PFNA and PFOS cause the same general types of hepatic toxicity. However, as summarized by Post *et al.* (2017), the hepatic effects of PFOS in rodents appear to be primarily PPAR- α independent (DWQI 2018), while hepatic effects of PFOA (DWQI, 2017) and PFNA (DWQI, 2015) involve substantial contributions from both PPAR- α dependent and independent processes. Likewise, while the developmental effects of PFOA, PFOS, and PFNA are generally similar, most, but not all, developmental effects of PFOA (reviewed in DWQI, 2017) and PFNA (reviewed in DWQI, 2015) in mice are PPAR- α dependent, but the developmental effects of PFOS (reviewed in DWQI, 2018) appear to be independent of PPAR- α .

Consideration of human data

The examples of mixtures assessments provided in the draft framework are based on the four PFAS (PFOA, PFOS, PFBS, GenX) that currently have final EPA Reference Doses (RfDs); all of these RfDs are based on animal data. However, the RfDs for PFOA and PFOS and the cancer slope factor for PFOA in the EPA’s draft MCLG documents are based on human data, and additional toxicity factors based on human data may be developed in the future for other PFAS. The Panel suggests that EPA consider how toxicity factors based on human data could be used in evaluations of PFAS mixtures, including for mixtures where toxicity factors for some PFAS are based on animal data and for other PFAS are based on human data.

Use of NAMs data in component-based mixtures approaches for PFAS

The potential use of data derived from new approach methodologies (NAMs; e.g., high throughput assays, read-across) for hazard identification and dose-response evaluation for PFAS mixtures is mentioned in several places in the draft mixtures framework (p. 12, 27, 34, 37, 52). The Panel agrees with the draft framework’s statement (p. 41) that the use of NAMs data would

allow for evaluation of toxicity of “data-poor PFAS” detected in environmental media that would not otherwise be considered.

However, current EPA risk assessment guidance does not provide for the use of NAMs data as the basis toxicity factors such as RfDs, and state environmental agencies generally follow EPA risk assessment guidance in developing health-based standards and guidance values for environmental contaminants. Therefore, EPA and states may face difficulties in justifying and implementing standards or guidance values (either chemical-specific or mixture-based) based on NAMs data for contaminants (PFAS or others) in drinking water or other environmental media. Regarding this issue, EPA stated at the SAB Panel meeting on December 16, 2021, that the agency does not plan to develop guidance for use of NAMs data to develop toxicity factors in the near future. EPA also stated that the use of NAMs in mixtures assessment is currently “quite abstract,” and that it is not expected that NAMs data will be used as the basis for standards or guidance values in the near future. They further clarified that that an approach based on NAMs data might be used to get a sense of whether PFAS detected in drinking water pose a risk in the absence of traditional toxicity data and that EPA hopes to develop case studies using NAMs data to evaluate the potential risk of PFAS mixtures. This clarification of how EPA envisions the use of NAMs data in PFAS mixtures assessments is not included in the draft EPA mixtures document, and the Panel recommends that it be added to the final framework.

Additionally, it is important to recognize that the potential use of NAMs data to address environmental contaminants that lack sufficient human or animal data for traditional toxicity factor (e.g., Reference Dose) development is not specific to PFAS mixtures assessment. This is a key concern for both chemical-by-chemical and mixtures assessment of PFAS and other contaminants. This issue has become especially important because chemical-specific toxicity factors cannot be developed for several PFAS (e.g., perfluoropentanoic acid, perfluoroheptanoic acid) that commonly occur in drinking water because there are no or virtually no data on their toxicity in animals or humans. EPA’s decision to minimize animal studies in its toxicology research has added to this issue, although a few recent EPA ORD animal studies have yielded high impact, key information on developmental effects of several PFAS of current concern. Examples of these high impact *in vivo* studies of PFAS of current concern are: GenX - Conley *et al.* (2019) and Conley *et al.* (2021); Nafion Byproduct 2 - Conley *et al.*, (2021); and mixtures of PFOA and PFOS (recent studies highlighted in the draft mixtures framework).

Development of toxicity factors for PFAS for which final EPA toxicity factors are not available

The draft EPA mixtures document (p. 33, last paragraph) states that toxicity values are needed to address PFAS (and other contaminants) for which final EPA toxicity factors have not been developed. The draft mixtures framework also notes that several states have developed toxicity factors for several PFAS for which there are no EPA toxicity factors (see Post, 2021). As noted in the draft EPA mixtures document, EPA has developed guidance for the derivation of subchronic and chronic oral RfDs, and most or all states follow this EPA guidance.

The SAB Panel agrees with EPA’s recommendations that toxicity values for PFAS should be developed by scientists with appropriate expertise and that their basis should be transparent.

However, the recommendation that such toxicity values "undergo independent peer review" does not appear to be appropriate for inclusion in the EPA mixtures document and replacing it with, for example, "toxicity values should be derived using peer-reviewed approaches and include opportunities for scientific input and review" may be more applicable. This recommendation is not specific to toxicity values used in mixtures assessments and would apply equally to toxicity values used in chemical-by-chemical approaches for addressing PFAS in drinking water or other media. It is important to recognize that each state has its own processes (established in legislation, regulation, or by policy) for development of such toxicity values, and that these processes may or may not include formal independent peer review. In fact, the Minnesota Department of Health oral toxicity values mentioned in the draft mixtures document for potential use in HI calculations (p. 33, first paragraph) did not undergo external peer review.

In some states, advisory bodies consisting of scientific experts develop toxicity values and recommend them to state environmental agencies. These toxicity value recommendations may be posted for public comment as drafts and revised as appropriate in response to the public comments before finalization. While such a process may not be considered to be a formal "independent peer review," it is a rigorous process that considers extensive scientific input from outside of the agency that will use the toxicity factor. A recommendation in the EPA mixtures document for "external peer review" of toxicity values developed by states could potentially be used as the basis for challenges to the validity of such state processes that may not include formal "external peer review." If such a recommendation is to be included in the EPA mixtures document, it is strongly suggested that it be broadened to recommend the opportunity for scientific input and review in general, rather than specifically "external peer review."

Recommendations

Overall recommendation: The SAB PFAS Review Panel supports dose additivity based on a common outcome, instead of a common mode of action as a health protective default assumption and does not propose another default approach. However, it is recommended that the uncertainties associated with this approach be more thoroughly and clearly presented along with information supporting this approach. Additionally, for clarity, the difference between a MOA and a health outcome should be defined.

The Panel recommends that when data clearly indicate interactions other than dose additivity, the approach indicated by the data should be used and that EPA reevaluate the default assumption of dose additivity as additional data become available.

Currently, studies that indicate a common mode of action and dose additivity for PFAS are reviewed in the draft framework. The Panel recommends that the discussion be expanded to include studies that do not indicate dose additivity and/or a common mode of action for PFAS.

The Panel recommends that EPA consider how toxicity factors based on human data could be used in evaluations of PFAS mixtures, including mixtures in which toxicity factors for some PFAS are based on animal data and toxicity factors for other PFAS are based on human data. The Panel recommends that EPA clarify its intended use of NAMs data in PFAS mixtures

assessments.

The Panel expressed concern regarding the requirement for "external peer review" of toxicity values developed by states and recommends that this phrase in the draft framework be broadened to recommend the need for scientific input and review in general.

Charge Question #2- Hazard Index Approach

Section 4.3 (Hazard Index, HI) of the framework demonstrates the application of a component-based mixture approach, based on dose addition, using available oral reference doses from completed EPA human health assessments, and hypothetical exposure information. The example calculations presented are primarily focused on four PFAS with finalized EPA Human Health Assessments: perfluorooctanoic acid (PFOA), perfluorooctane sulfonic acid (PFOS), perfluorobutane sulfonic acid (PFBS), and hexafluoropropylene oxide (HFPO) dimer acid and HFPO dimer acid ammonium salt (referred to as "GenX chemicals").

A. Please provide specific feedback on whether the HI approach is a reasonable methodology for indicating potential risk associated with mixtures of PFAS. If not, please provide an alternative.

The Panel agrees with the use of Hazard Index (HI) as a screening method and decision-making tool (see "limitations" discussion below).

In general, the screening level Hazard Index (HI) approach, in which Reference Values (RfVs) for the mixture components are used regardless of the effect on which the RfVs are based, is appropriate for initial screening of whether exposure to a mixture of PFAS poses a potential risk that should be further evaluated. Toxicological studies to inform human health risk assessment are lacking for most members of the large class of PFAS, and mixtures of PFAS that commonly occur in environmental media, overall. For these reasons, the HI methodology is a reasonable approach for estimating the potential aggregate health hazards associated with the occurrence of chemical mixtures in environmental media. The HI is an approach based on dose additivity (DA) that has been validated and used by EPA. The HI does not provide quantitative risk estimates (i.e., probabilities) for mixtures, nor does it provide an estimate of the magnitude of a specific toxicity. This approach is mathematically straightforward and may readily identify mixtures of potential toxicological concern, as well as identify chemicals that drive the toxicity within a given mixture. As described in the draft framework, this approach has advantages and limitations that were adequately described.

The Panel also found that the approaches described in the draft framework would be better described and used as a menu-based approach rather than a tiered one. Given the Agency's desire to support fit-for-purpose approaches, not every PFAS mixture scenario will be one that warrants a tiered or hierarchical approach. In some instances, an HI or target-organ-specific hazard indices (TOSHI) might provide enough information for decision-making about PFAS (or other chemicals) contamination in drinking water (or other media). Tiered approaches that require increasingly complex information before reaching a final decision point can be extremely challenging for data poor chemicals such as PFAS. Data gaps identified in a such tiered

methodologies could result in a bottleneck through which these chemicals may never emerge. However, it is important to recognize that the information provided for decision-making by HI and TOSHI (e.g., that a mixture poses a potential risk) differ from the quantitative toxicity information provided by more refined approaches such as relative potency factor (RPF).

B. *Please provide specific feedback on whether the proposed HI methodologies in the framework are scientifically supported for PFAS mixture risk assessment.*

There are several challenges and considerations relevant for PFAS mixture risk assessment. These challenges are articulated below.

Lack of toxicology data

Even for the Screening Level HI calculations, the number of PFAS with available toxicological assessments remains limited, and many users will not have the ability to derive them using the methods outlined in the draft framework. Additional guidance will likely be necessary for most users.

Use of TOSHI approach

The TOSHI approach necessitates endpoint/health effect-specific reference values, not just overall reference values. Therefore, the draft framework should be clearer in explaining that endpoint/health effect-specific reference values must be developed for individual PFAS. The TOSHI approach presents additional robustness compared to the Screening Level HI given the identification of human health/toxicity values that are effect/endpoint specific. However, the framework appears to classify both the Screening Level HI and the TOSHI approach as being equivalent Tier 1 methods that should lead to a more robust Tier 2 approach (i.e., RPF). The TOSHI approach may merit consideration to be classified as a higher tier method compared to the Screening Level HI method for decision making purposes. This may also reflect current and future practices amongst states and others. Currently, both Screening Level HI and TOSHI are classified as Tier 1 methods that should be followed up by a Tier 2 method, such as RPF.

Consideration of probabilistic methods for HI/TOSHI calculations to estimate risk.

The National Academies of Sciences, Engineering, and Medicine (NASEM, 2021) review of the 2020 IRIS Handbook recently endorsed the IRIS program's development of probabilistic risk-specific doses to replace traditional deterministic reference values. In the future, EPA should consider the extent to which using the corresponding "probabilistic RfD" or "risk-specific doses" would change the proposed HI/TOSHI approach, or whether such probabilistic reference values can be used as direct replacements for the traditional RfD in HI/TOSHI calculations. It should be noted that the risk-specific doses derived from these methods provide actual estimates of risk in the form of population incidence (e.g., 1% of the population) for a particular magnitude of effect (e.g., 5% change in ALT) at a particular confidence level (e.g., 95% confidence), and thus provide more than "indicating potential risk" and are more akin to "estimating risk."

Challenges with implementation

An HI and TOSHI do not provide quantitative estimates of risks associated with PFAS mixtures in a given exposure (see discussion of probabilistic methods above that could provide risk estimates). Nonetheless, these approaches could be useful for categorizing a specific mixture as to its potential hazard. Additionally, HI/TOSHI estimates need to be interpreted with caution in that different mixture exposure scenarios that contain the same chemicals may result in the derivation of identical His/TOSHIs. However, due to factors specific to each exposure scenario, they may not necessarily exhibit the same potential for causing adverse health effects.

Another disadvantage of the HI/TOSHI approach for specific exposure scenarios (and environmental media) is that it requires derivation of a health-based, media-specific concentrations (e.g., drinking water Health Advisory or MCLG) in addition to chemical-specific toxicity values (e.g., Reference Doses). As shown in Table 4-3 (p. 39) of the draft EPA mixtures document, development of health-based water concentrations (HBWCs) requires chemical-specific toxicity values) and chemical-specific exposure assumptions (e.g., ingestion rates, Relative Source Contribution factors). Additionally, HBWCs may apply to different exposure durations (i.e., short-term, subchronic, chronic). The draft framework should consider whether it is appropriate to use HBWCs based on different exposure assumptions and/or different exposure durations in HI evaluation of PFAS mixtures. For example, the HBWCs used in the examples of the HI approach (Section 4 of the draft EPA mixtures document) are the U.S. EPA (2016) Health Advisories (Has) for PFOA and PFOS (USEPA, 2016a; USEPA, 2016b). As shown in Table 4-3, the PFOA and PFOS Has are based on the drinking water ingestion rate for lactating women which is higher than the default adult ingestion rate. The ingestion rate for lactating women was selected because PFOA and PFOS are transferred to breastmilk, and exposure to PFOA and PFOS in breastfed infants (via maternal consumption of PFOA/PFOS-contaminated drinking water) is higher than in infants who consume formula prepared with the contaminated water or older individuals. However, ingestion rates for subgroups other than lactating women (e.g., infants, children, default adults) may be appropriate for HBWCs for other PFAS. For example, the ingestion rate for lactating women is not likely to be appropriate for HBWCs for PFBS or GenX, since there is no information to indicate that PFBS or GenX are present in breastmilk. Additionally, the U.S. EPA (2016a; 2016b) Health Advisories are stated to apply to both short-term (weeks to months) and chronic exposures, while HBWCs for other PFAS might apply to different exposure duration(s). As above, EPA should consider these issues in developing the HI methodologies for PFAS mixtures that use HBWCs.

In the example in Table 4-4 of the draft framework, the individual concentrations for PFOA and PFOS are of 20 ng/L for each which is below the HBWCs of 70 ng/L for these chemicals and the combined concentration of PFOA and PFOS is also below 70 ng/L. It is therefore not unexpected that the HI is below 1 for the combined concentration. In the example in Table 4-5 of the draft framework, the individual concentrations of PFOA and PFOS of 400 ng/L exceed the HBWC of 70 ng/L, so it is not unexpected that the HI for the combined concentration (and for each individual PFAS) exceeds 1. It would be useful to provide an additional example in which the concentration of each individual PFAS is below its HBWC (e.g., the health advisory (HA)),

yet the combined HI that considers both PFAS exceeds 1. For example, 40 ng/L for PFOA and 50 ng/L for PFOS.

Limitations

There are some limitations and potential complications in terms of the intended users such as states and public water systems applying this framework in the context of implementing the Safe Drinking Water Act. Additional clarity and guidance from EPA will be helpful in mitigating any inadvertent uncertainties caused by the issuance of this framework in a final form. More details on the intent, purpose, and potential applications of this framework by stakeholders such as states, public water systems and others will be helpful. For instance, some states that have promulgated either regulatory or guidance values for PFAS are using a mixtures-based approach for the specific combination of PFAS compounds prevalent in the state. Methods analogous to those classified by EPA as ‘Screening Level’ or ‘Tier 1’ in the framework are potentially being used by states in a decision-making capacity. Issuance of this framework without recognition of that fact may create confusion for public water supplies and risk communication challenges for the public. Additionally, should EPA promulgate National Primary Drinking Water Regulations (NPDWRs) for PFOS and PFOA as proposed, it should be clarified how those will factor into a mixtures approach for making decisions at public water systems

The equivalency of HI calculations using the different categories of toxicity assessment information available as presented in Table 4-2 should be clarified. It would also be beneficial to outline the validity of, and procedures for, calculating the HI should the mixture present include PFAS compounds with varying levels of information available, i.e., fall in different rows of Table 4-2.

Recommendations

The Panel recommends that EPA consider using a menu-based approach rather than a tiered approach as described in the draft Mixtures document. Tiered approaches that require increasingly complex information before reaching a final decision point can be extremely challenging for data poor chemicals such as PFAS.

In the future, EPA should consider the extent to which using the corresponding “probabilistic RfD” or “risk-specific doses” would change the proposed HI/TOSHI approach, or whether such probabilistic reference values can be used as direct replacements for the traditional RfD in HI/TOSHI calculations.

The Panel recommends that EPA provide additional clarity and guidance for implementing the framework to mitigate any inadvertent uncertainties, such as use different ingestion rates when basing mixtures assessments on HBWCs.

Charge Question # 3- Relative Potency Factor

*Section 4.4 (**Relative Potency Factor, RPF**) of the framework demonstrates the application of a component-based mixture approach, based on dose addition, using available dose-response information (i.e., points-of-departure) from completed EPA human health assessments, and hypothetical exposure information. The example RPFs and corresponding Index Chemical Equivalent Concentration (ICEC) calculations presented are primarily focused on four PFAS with finalized EPA Human Health Assessments: PFOA, PFOS, PFBS, and GenX chemicals.*

A. Please provide specific feedback on whether the RPF approach is a reasonable methodology for estimating risk associated with mixtures of PFAS. If not, please provide an alternative.

B. Please provide specific feedback on whether the proposed RPF methodology in the framework is scientifically supported for PFAS mixture risk assessment.

The draft EPA mixtures document describes three different approaches for assessing mixtures. The Hazard Index (HI), Target-Organ Specific HI (TOSHI), and Benchmark Modeling (BMD) that lead to either the derivation of Relative Potency Factors (RPFs) and Internal Chemical Equivalency Concentrations (ICECs) or a mixture BMD. RPFs quantify relative potencies of substances with respect to an effect and can be used to express combined exposures of multiple substances in terms of the exposure value of the chosen index substance (i.e., as index substance equivalents) (MCRA 9).

The Panel considers the RPF approach a reasonable methodology for assessing mixtures. The Panel noted that the RPF approach is a more data intensive approach, as compared to the Hazard Index methods, which is likely to see a greater application for PFAS. They expressed concern that there are many PFAS with little or no data and an approach is needed to address mixtures where comprehensive datasets do not exist. To summarize, the Panel agrees that the RPF approach is a reasonable methodology for estimating risk associated with mixtures of PFAS, and did not suggest an alternative methodology. The Panel agreed that the framework should reconsider the tiered approach that is presented in Figure 4-1. They also noted that New Approach Methods (NAMs) may be useful in filling data gaps for some PFAS given the large number of these substances that lack data.

The Panel concluded that the framework needs further elaboration and clarification before it can be implemented. Firstly, the draft framework delves into high level details about the various methodologies proposed without substantial discussion of the methodologies (e.g., HI, TOSHI, RPF, BMD) until later in the framework. Secondly, the Panel noted that it would be helpful to have guidance about the types of data sets are most applicable for each approach. Alternatively, an additional figure/figures could be included with a flow diagram as provided in Figure 4-1. Beginning with a particular type of data set and then providing guidance on which methodology to use. The Panel also agreed that the draft framework should re-evaluate the tiered approach as they questioned whether HI or TOSHI is needed before moving on to RPF and suggested using a menu of options based on the available data instead. Overall, there was agreement that removal of tiers would enhance the framework.

The Panel suggested that it would be helpful if the draft framework provided clarification regarding the conceptual differences between the TOSHI approach and the RPF approach, since both are based on health effect-specific values (i.e., RfVs or RPFs) for the individual PFAS in the PFAS mixture. Moreover, the Panel agreed that the framework should also summarize when the TOSHI and RPF approach will give essentially the same answer (e.g., when the ratio of the POD values used to calculate the RPFs is equal to the ratio of the endpoint-specific RfD values used to calculate the HI), and the extent to which consistency is appropriate or unnecessary (e.g., if one is supposed to be more “conservative” for screening, rather than more “predictive.”). Overall, the Panel concluded that more discussion and comparison of approaches, as well as when they converge, is needed. As noted below in Charge Question #4, an example was provided demonstrating that the RPF and mixture BMD approaches can be very similar or even equivalent, indicating that differences between them should not be exaggerated. The draft framework should provide further discussion and explanation focusing on this concern. The Panel raised similar concern about the differences between TOSHI and RPF give similar answer. Lastly, the framework should give guidance on the approaches, and which approach is preferable- a more conservative versus predicative one.

The Panel also agreed that the framework should increase the discussion, rationale, and justification with regard to Index Chemical (IC) selections. The Panel suggested that the framework would be strengthened by providing a flowchart or process in which the IC can be determined. An example appreciated by the Panel was the comparison of PFOA to PFOS as the IC versus PFOS to PFA.

Lastly, the RPF approach is based on the assumption of dose additivity and use of a common health effect/toxicity endpoint as a surrogate for a common MOA, as discussed in Charge Question 1 above. As such, the comments on the scientific basis of these assumptions from Charge Question #1 apply here as well. The scientific basis for the RPF approach presented in the draft mixtures framework is strengthened by the use of PODs from animal studies that are based on human equivalent doses (HEDs) rather than administered doses. In contrast, the PFAS RPFs based on BMDs for a 5% increase in relative liver weight from subchronic exposure to male rats developed by Bil *et al.* (2020), which are being used to address PFAS mixtures by some European environmental authorities, are based on administered dose and do not consider differences among PFAS regarding animal-to-human toxicokinetic extrapolation.

Recommendations

The Panel recommends that the framework provide further elaboration and clarification before it can be implemented, including providing guidance about the types of data sets that are most applicable for each approach.

The Panel recommends that the draft framework provide clarification regarding the conceptual similarities and differences between the TOSHI approach and the RPF approach, since both are based on health effect-specific values (i.e., RfVs or RPFs) for the individual PFAS in the PFAS mixture. Therefore, more discussion and comparison of approaches, as well as when they converge, is needed.

The Panel recommends that the framework increase the discussion, rationale, and justification with regard to Index Chemical (IC) selections.

Charge Question #4- Mixture BMD

Section 4.5 (Mixture BMD) of the framework demonstrates the application of a component-based mixture approach using established EPA dose-response modeling (i.e., benchmark dose, BMD) of hypothetical PFAS dose- response data, and hypothetical exposure information.

A. Please provide specific feedback on whether the Mixture BMD approach is a reasonable methodology for estimating what is in essence a mixture-based point-of-departure. If not, please provide an alternative.

The proposed method employs a dose-additive model-based calculation of a mixture BMD based on a defined benchmark response (e.g., ED10) for a PFAS mixture with a specific mixing-ratio of component chemicals, as dose additivity has been viewed as the most appropriate model for estimating combined effects of “toxicologically similar” compounds.

In general, the Panel agreed that the Mixture BMD approach is a reasonable methodology for estimating a mixture-based POD.

Relationship to other approaches

While the mixture BMD approach was deemed reasonable, some caveats were identified. As with the RPF approach, the framework should also discuss scenarios in which the TOSHI and Mixture BMD approach will give essentially the same answer. That is, when the ratio of the BMD values used to calculate the mixture BMD is equal to the ratio of the endpoint-specific RfD values used to calculate the HI. Also, the extent to which this consistency is appropriate or inappropriate should be clarified (e.g., if one approach is intended to be more “conservative” for screening, rather than more “predictive.”).

Further, the RPF and mixture BMD approaches appear to be very similar or even equivalent; differences between them should not be exaggerated. Both approaches appear to be a summary measure of the toxicity of a mixture, $ICEC_{MIX}$ for the RPF approach and t_{add} for the mixture BMD approach. Both approaches are weighted sums of the component concentrations, with weights proportional to some measure of toxicity (e.g., inverse of BMD or of ED10).

Combining eqns. (4.2) and (4.3) implies

$$ICEC_{MIX} = \sum_j d_j \frac{ED10_{IC}}{ED10_j} = ED10_{IC} \sum_j \frac{d_j}{ED10_j}. \quad (1)$$

Taking reciprocals in eqn. (4.5) yields

$$\frac{1}{t_{add}} = \sum_i \frac{a_i}{BMD_i} \quad (2)$$

Comparing these results shows that $ICEC_{MIX}$ and $(t_{add})^{-1}$ differ only in inessential details and are essentially proportional to one another, as follows: It is first noted that the RPF approach can use any common toxicity value, e.g., one can replace ED10 with BMD in equation (1). Second, d_i (the “component chemical’s concentration”) and a_i (“the fixed proportions of the component PFAS in the mixture”) are either identical or strictly proportional to each other. Third, $ICEC_{MIX}$ includes a constant proportionality factor ($ED10_{IC}$).

Having constructed $ICEC_{MIX}$ and t_{add} , one can presumably use them to evaluate a risk in analogous ways. The summary t_{add} can be used as a BMD, from which one can calculate a hazard index or use it as a POD from which to extrapolate a dose-response function. Similarly, one can divide $ICEC_{MIX}$ by $ED10_{IC}$ to calculate a hazard index or use it in a dose-response function for the index chemical as in equation (4.4).

Given these mathematical correspondences, EPA should consider revising the discussion of these two approaches to present them as essentially the same (or highlighting any essential differences), and perhaps also merging them into a single section.

Utility of the mixture BMD

The draft framework did not clearly present the practical utility of this approach as compared to other mixture approaches, and the Panel found it difficult to envision situations in which the mixture BMD was advantageous. The output of this approach is a BMD in units of mg/kg/day for the total concentration of a mixture of specific PFAS in specific proportions. At the SAB meeting on December 16, 2021, EPA stated that a Mixture BMD could be used to characterize a unique site or exposure and that it is applicable when it is fairly certain that the composition of the mixture is relatively stable. However, it is unclear what benchmark the Mixture BMD could be compared to in order to determine whether or not there is a potential risk from a mixture of PFAS in drinking water or other environmental media. The method as described in the draft mixtures document is based on endpoints that, while critical, may prove difficult to obtain for many environmental chemicals – especially the thousands of PFAS known to exist. While the inclusion of the possibility of NAMs filling data gaps was suggested, for thyroid and developmental endpoints, current NAMs are quite limited and thus could be a limiting factor in the use of this method. The proposed approach would also benefit from additional information on how the method will be applied in practice, e.g., whether for specific mixtures, i.e., those that are found in a specific location or water system or whether the method is fit-for-purpose enough to help water system operators or regulators determine if a system is in excess of the MCLG (and eventual MCL). Development of additional case studies that highlight how this method would work for a real-world sample (rather than a hypothetical case) and how it would work with data poor chemicals would be helpful in establishing both scientific confidence in the method and evaluation of whether it is fit for its intended purpose.

1
2 *B. Please provide specific feedback on whether the proposed Mixture BMD methodology*
3 *in the framework is scientifically supported for PFAS mixture riskassessment.*
4

5 In general, the Panel agreed that the approach is scientifically supported for PFAS mixture risk
6 assessment, and that both its criteria for application and its potential limitations are well
7 described. Throughout the draft framework for PFAS, the EPA clearly explained the BMD
8 process and approach and appear to have followed the basic recommendations in the EPA's
9 *Supplementary Guidance for Conducting Health Risk Assessment of Chemical Mixtures* (2000).
10

11 The Panel noted that an advantage of this approach is that only PODs (NOAELs, LOAELs,
12 BMDs) rather than RfVs (RfDs, HBWCs) are needed. However, the RPF approach is also based
13 on PODs, rather than HBWCs or RfDs. In the RPF approach, the PODs are based on human
14 equivalent doses (HEDs) rather than administered doses. However, the use of HEDs does not
15 appear to be shown in the Mixture BMD approach. The use of PODs based on HEDs is
16 recommended, and it should be clarified that PODs based on HEDs should be used in the
17 Mixture BMD approach. Case studies that illustrate these points, using real-world scenarios,
18 could be useful in highlighting this change.
19

20 **Recommendations**

21
22 The Panel recommends that additional information on how the proposed Mixtures BMD
23 approach will be applied in practice be provided.
24

25 The Panel recommends that PODs based on HEDs be used in the Mixture BMD approach and
26 EPA should clarify this with case studies that illustrate these points, using real-world scenarios to
27 highlight this change.
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SECTION III- Benefits from CVD reduction

EPA's draft Analysis of Cardiovascular Disease (CVD) Risk Reduction as a Result of Reduced PFOA and PFOS Exposure in Drinking Water

Overall Charge Question

EPA is seeking SAB evaluation on the extent to which the approach to estimating reductions in CVD risk associated with reductions in exposure to PFOA and PFOS in drinking water is scientifically supported and clearly described.

The approach is quite reasonable overall given that perhaps the most well-established effect of PFAS exposure is elevated serum cholesterol. The threshold for assessing the benefits of reducing PFAS exposure levels is indicated to only require “a meaningful opportunity for health risk reduction” and that does seem to be the case here. Therefore, examining the sequence from regulation and reduction of PFAS in drinking water to changes in serum PFAS to changes in serum cholesterol to changes in rate of cardiovascular events provides a reasonable basis for estimating the cardiovascular benefits of the regulation. In the draft CVD document, the Panel recommends that EPA provide a clearer rationale and list the main assumptions at the outset before launching into the considerable detail that follows. EPA should specify additional assumptions and explanations, perhaps by expanding Section 2 of the report.

The Panel expressed a concern about the apparent discrepancy between this document's focus on CVD risk, and the draft MCLG documents' conclusions that the evidence of CVD was not sufficient to form the basis of a RfD. As discussed in the Panel's review of the draft MCLG document, the Panel viewed the most consistent epidemiological associations with PFOA and PFOS to include decreased immune system response, decreased fetal growth (e.g., decreased birthweight), increased serum lipids, and increased liver enzymes (particularly ALT), with none of these four endpoints being substantially stronger than the other. Overall, the Panel recommends more discussion as to the rationale for selecting this particular endpoint for risk reduction analysis (e.g., strengthening of the hazard conclusion with respect to PFOA or PFOS, availability of dose-response data from which to derive a dose-response function or risk-specific dose estimates, strengthening of data connecting changes in biomarker to changes in morbidity or mortality, and availability of data for monetizing benefits).

Specifically, with respect to CVD, the assumption is that a shift in cholesterol resulting from PFAS exposure will have the same impact on cardiovascular disease that cholesterol levels based on natural levels or use of cholesterol lowering medications have had. However, the epidemiologic literature that provides strong support for an effect of PFAS on cholesterol does not provide support for an effect of PFAS on the risk of cardiovascular disease. This does not negate the value of the approach, but the Panel suggests the inclusion in the draft CVD document of an acknowledgement that the approach follows the pathway that links cholesterol to cardiovascular events rather than looking at the reported effects of PFAS directly on cardiovascular disease.

The temporal sequence of events is addressed at many points in the technical details, but the Panel suggests that the draft analysis start with an overview and orientation to help the reader follow what comes next. While the population serum PFAS levels would be altered over time as PFAS levels decrease in drinking water, the half-life data indicate that this would not be instantaneous. The EPA draft CVD document states that with that altered serum PFAS level an instantaneous shift in serum cholesterol would result. The altered serum cholesterol level would then result in changes in the risk of cardiovascular events (e.g., the number of such events occurring per year).

The Panel concluded that performing a sensitivity analysis for each step from beginning to end is worth considering while recognizing the need to manage the complexity and volume of results. Perhaps in addition to the “most likely” approach that was used, a chain of assumptions that lead to “best case” (maximum benefit) and “worst case” (minimum benefit) would be worth considering.

Additionally, while not directly relevant to the modeling of cholesterol and CVD risk, this analysis raises the question as to what methodologies will be used for estimating health benefits for other endpoints (i.e., those with candidate RfD values in the MCLG document). For instance, it seems that decreased birth weight (as a continuous endpoint or as an increase in the percentage of low-birth-weight infants) would be a good candidate for estimates of risk reduction. Some studies in the economic literature that examine the economic consequences of decreased birth weight include Almond (2005), Black *et al.* (2007), Almond *et al.* (2010), Barreca *et al.* (2011), Fletcher (2011), Figlio *et al.* (2014), Bharadwaj *et al.* (2013), and Chyn *et al.* (2021). If EPA were to pursue monetizing this endpoint, we recommend a more thorough literature review for this, as well as other endpoints.

Recommendations

The Panel recommends that EPA provide a clearer rationale and list the main assumptions at the outset before launching into the considerable detail that follows.

The Panel recommends more discussion as to the rationale for selecting this particular endpoint for risk reduction analysis (e.g., strengthening of the hazard conclusion with respect to PFOA or PFOS, availability of dose-response data from which to derive a dose-response function or risk-specific dose estimates, strengthening of data connecting changes in biomarker to changes in morbidity or mortality, and availability of data for monetizing benefits), as well as considering risk reduction analyses for other endpoints.

While recognizing the need to manage the complexity and volume of results, the Panel recommends performing a sensitivity analysis for each step from beginning to end.

Charge Question #1- EPA's Meta-Analysis

Section 4.2 presents EPA's meta-analysis for the total cholesterol dose-response function.

- i. *Please provide specific feedback on the extent to which the study selection criteria, the identified studies, and the methodological approach of the meta-analysis are complete and capture up to date scientific literature.*
- ii. *To inform the CVD risk reduction analysis for those ages 40-89 using the ASCVD risk model, EPA used a meta-analysis approach for the total cholesterol dose-response function. Please provide specific feedback on the extent to which this approach is reasonable for this application, or whether using a single dose-response study (e.g. Dong et al., 2019) selected in the analysis of cholesterol impacts in the "Proposed Approaches for Deriving Maximum Contaminant Level Goals for PFOA and PFOS in Drinking Water" would add additional strengths for the CVD risk reduction application.*

The approach to identifying and managing the literature on PFAS and cholesterol is reasonable for this purpose, with little basis for more esoteric or complex approaches. What is needed, however, is some sense of the impact had other approaches been used, setting the stage for a sensitivity analysis. If, in fact, a variety of reasonable alternative options would all generate roughly the same predictions, then this would be sufficient to put aside concerns with the specific approach used. If reasonable alternatives would lead to meaningfully different estimates, it would be useful to carry through one or more alternatives to the final results in a sensitivity analysis.

The meta-analysis study selection criteria are well reasoned, and EPA identified 14 relevant studies for analysis. One interesting study (Fitz-Simon *et al.*, 2013) was not included probably due to its different study design of modeling prospective cholesterol level changes by PFAS level changes in the C8 Health Study, but it is really relevant to the topic of the meta-analysis. Even though its design may not make it easy to be integrated into the meta-analysis, the interpretation of the meta-analysis findings should consider this study in the array of pertinent results.

Because the NHANES has been repeatedly analyzed for the association between PFAS and total cholesterol levels, using meta-analysis may not yield additional value compared with the approach Dong *et al.* used in the 2019 publication. If the results were from different study populations, meta-analysis results would provide a useful method to aggregate across these populations.

There are a series of related concerns regarding the basis for exclusion of studies from consideration. The criterion of "special populations" requires a clearer explanation and reconsideration. The Panel recommends that EPA list the studies that were excluded from the meta-analysis and provide a brief description of these studies and why each was excluded. EPA mentions that studies performed on specific population subsets were not considered for inclusion in the meta-analysis, but the basis for exclusion other than occupational populations is unclear. For example, what is the rationale for determining that the Steenland *et al.* 2009 study (C8 cohort) is not a "special population" subset? Similarly, studies of the Inuit Population of

1 Nunavik (Château-Degat *et al.*, 2010) and Chinese Male Adults (Yang *et al.*, 2018) were
2 included but might not have been.
3

4 Study quality as the basis for inclusion and exclusion is likewise a relevant consideration that
5 needs to be carefully defined and consistently applied. In the Appendix, EPA mentions that all
6 of the studies in the meta-analysis, except one (Lin *et al.* 2019) are cross-sectional with various
7 methodological limitations. It would be helpful to know if the quality of the cross-sectional
8 studies is viewed any differently than Lin *et al.* (2019) and what the implications of the analysis
9 are when using studies of varying quality, if any. More details are needed regarding the protocol
10 for risk of bias assessment/confidence rankings and whether that approach was used before
11 considering the weight of the evidence and/or carrying studies forward for meta-analysis. The
12 evidence synthesis protocol should also include a “tiered” approach to evaluate whether results
13 or conclusions change based on varied decisions about inclusion of high, medium and low
14 confidence studies across various study design domains (as referenced in the overall discussion
15 of sensitivity analyses above). As discussed in more detail in Section I (MCLG documents),
16 excluding studies automatically for insufficient adjustment for socioeconomic status may also
17 not be warranted.
18

19 In addition to variation due to selection of studies, the impact of modeling approach also
20 warrants a sensitivity analysis. For example, the linear untransformed models yield by far the
21 highest slopes, especially for PFOA which calls for careful interpretation. For extraction of
22 slope values for total cholesterol (TC) and high-density lipoprotein cholesterol (HDLC), Section
23 A.2.1 (pg. 6) states that when multiple models with different confounders were reported within a
24 single study, *either* the most adjusted results or the main model results were selected.
25

26 In the slope estimation, the associations for HDLC and PFOA and PFOS were positive, albeit not
27 statistically significant, which may not warrant exclusion of HDLC from consideration in the
28 CVD risk reduction analysis. Selecting endpoints solely on the basis of having statistically
29 significant positive effects may exclude meaningful associations that are imprecise. The Panel
30 concluded that further explanation of the basis for excluding this indicator from detailed
31 consideration would be helpful. For instance, the causal connection between pharmacologically-
32 induced changes in HDLC and changes in CVD morbidity or mortality is less clear than for TC.
33

34 Finally, the Panel agreed that the approach for estimating the dose-response function using the
35 Atherosclerotic Cardiovascular Disease (ASCVD) risk assessment tool is likely a reasonable
36 choice. It was developed by leading cardiovascular disease researchers, it is widely used despite
37 some limitations, and there is no reason to invent a new tool for the purposes of this exercise. In
38 fact, it seems the instrument was developed for exactly this purpose, to estimate the impact of
39 interventions including modifying cholesterol levels, and the reduction of PFAS in drinking
40 water is one of the ways this might be accomplished. For conducting sensitivity analysis, we
41 would not expect that other models would yield notably different results, or even if they would,
42 that the others are somehow more accurate.
43
44
45

Recommendations

The recommendations from Section I (MCLG documents) with respect to evidence identification and evaluation should also be applied here where applicable.

The Panel recommends that EPA list the studies that were excluded from the meta-analysis and provide a brief description of these studies and why each was excluded.

The Panel recommends that the evidence synthesis protocol also include a “tiered” approach to evaluate whether results or conclusions change based on varied decisions about inclusion of high, medium and low confidence studies across various study design domains.

Charge Question #2- EPA’s Life Table Approach

Section 5.1 presents EPA’s life table approach methodology.

i. Please comment on the extent to which this analysis is scientifically supported and clearly described. To the extent improvements are suggested, please provide specific changes that are implementable in a U.S. national-level benefits analysis with readily available data.

Overall, the application of the life table methodology to evaluate CVD risk reduction from treatment of PFOS/PFOA is reasonable and the methodology is generally well-described.

As the draft EPA CVD document highlights, this approach has been used by the EPA previously (e.g., to evaluate the benefits and costs of the 2015 Steam Electric Rule, US EPA 2015 and the Clean Air Act, US EPA 2011). When discussing prior applications (p. 15), the Panel recommends that EPA describe how the current application of the life table methodology differs in the use of prevalence statistics and other key input data and assumptions. Presumably, the EPA will use the resulting estimates to inform the economic analysis of PFOA and PFOS treatment in public water systems. The Panel concluded that an outline is needed explaining how the CVD risk reductions captured using this approach will be monetized, as well as clarifying the differences from previous EPA analyses that have monetized cardiovascular impacts. For example, in the calculation of Clean Air Act benefits (US EPA, 2011), the cardiovascular impacts monetized include premature mortality, myocardial infarction, and cardiovascular hospital admissions, whereas the current application examines fatal and non-fatal myocardial infarction, fatal and non-fatal ischemic stroke, and other coronary heart disease mortality.

An advantage of the life table approach is that it takes into consideration the timing of the treatment and the aging of the population, which captures the impact of treatment on partially treated populations (i.e., those born before treatment occurred). The proposed methodology, however, involves many assumptions and modeling decisions that may affect the estimates of the mortality/morbidity impacts, such as excluding individuals with pre-existing conditions and tracking post-acute CVD mortality for up to five-years after a CVD incident. These modeling assumptions are likely to omit components of CVD benefits. The Panel recommends that these assumptions be clearly listed (e.g., in Table 7).

While the overall approach in Section 5.1 was clear, Table 3 could benefit from reorganization to avoid confusion. Specific suggestions for Table 3 include the following: (1) add two additional columns that separate CVD and Non-CVD calculations under “Baseline Calculations” and “Treatment Scenario Calculations” or split Table 3 into calculations specific to baseline vs. treatment scenarios, (2) “linearize” steps in Table 3, as the steps are not consecutively ordered, (3) add equation labels from Appendix B (B.3.2, B.3.3, B.3.4) in the Table 3 steps, and, where possible, map steps in Table 3 to steps in Figure 3.

Additionally, a list of specific areas in the EPA draft CVD document and the EPA appendix that need clarification/revision has been provided in Appendix A.

Recommendations

The Panel recommends that EPA describe how the current application of the life table methodology differs in the use of prevalence statistics and other key input data and assumptions from prior applications.

The Panel recommends that assumptions and modeling decisions in the proposed methodology, that may affect the estimates of the mortality/morbidity impacts, such as excluding individuals with pre-existing conditions and tracking post-acute CVD mortality for up to five-years after a CVD incident be clearly listed (e.g., in Table 7).

Charge Question #3- ASCVD Risk Model

Section 5.2 presents EPA’s application of the atherosclerotic cardiovascular disease (ASCVD) risk model used to estimate the probability of hard CVD events corresponding to total cholesterol changes.

i. Please comment on the scientific validity of the ASCVD model application for estimating the probability of first time CVD events in various sub-populations and the extent to which it is clearly described.

The ASCVD pooled cohort equation (PCE) risk model is a scientifically valid approach to estimating the probability of first CVD events. A limitation is that it is estimated only for non-Hispanic white and Black populations.

Many studies have examined the discrimination and calibration of the PCE risk model, and both under-estimation and over-estimation of risk have been demonstrated depending on the characteristics of the study sub-population, in particular in relation to age, race and socioeconomic factors. For example, epidemiologic data have shown that ASCVD risk is overestimated among Black and white adults with less social deprivation, defined on the basis of income, education and living alone (Colantonio *et al.*, 2017). Among those with more social deprivation, the PCE has good calibration or underestimates CVD risk. Thus, the Panel

1 recommends that EPA provide further discussion of the accuracy of the model predictions in
2 sub-groups with varying levels of social deprivation.

3
4 The PCE was derived from cohort data among individuals aged 40-79 years and the model does
5 not allow for changes in risk associations with increasing age (i.e., it assigns fixed weights to
6 each risk factor regardless of age). A recent study (Dalton *et al.*, 2020) among individuals aged
7 65+ demonstrated poor performance of the ASCVD model in predicting cardiovascular events in
8 this older population, with the exception of white males aged 65-74. Of particular relevance,
9 associations with systolic blood pressure, total cholesterol, and diabetes weakened as a function
10 of age.

11
12 Because the ASCVD model is estimated only for non-Hispanic whites and non-Hispanic Blacks,
13 EPA needs to choose how to estimate risk for other populations. Although Goff *et al.* (2014)
14 suggested using the coefficients for non-Hispanic whites, EPA found that projections using the
15 coefficients for non-Hispanic Blacks provide a closer match to population prevalence and
16 mortality for Hispanic and non-Hispanic other populations, and appropriately adopted these
17 coefficients in its analysis.

18
19 The accuracy of the ASCVD model seems modest, even for the populations for which it is
20 estimated. Table B-12: Summary of ASCVD Model Validation reports the population-weighted
21 average (over sex and race/ethnicity categories) of the absolute value of the proportional
22 deviation of reported incidence from modeled incidence. For females, the value of this statistic is
23 2.00 for non-Hispanic whites and 1.37 for non-Hispanic Blacks. For subpopulations to which the
24 model was not fitted (Hispanic and non-Hispanic other), the statistic is better, between 0.9 and
25 1.5. For males, model accuracy is substantially better, with the statistic ranging between about
26 0.2 and 0.6 for all population subgroups. The Panel concluded that the accuracy of the model
27 predictions deserves more discussion. Moreover, it is important to test and report the average
28 bias between model predictions and population incidence, overall and for population subgroups,
29 in addition to the measure of spread.

30
31 *ii. Please comment on whether EPA's approach and assumption of a uniform first*
32 *CVD event hazard distribution over the 10-year period is sufficiently robust given*
33 *current data sources and literature. If additional distributional sources of*
34 *information are suggested, please provide specific citations/sources for EPA's*
35 *consideration.*

36
37 The ASCVD model predicts the probability of a first CVD event in the following 10 years. EPA
38 estimates the annual risk as the constant yielding the same 10-year risk (i.e., $(1 - x)^{10} = (1 - y)$
39 where x is the annual risk and y is the 10-year risk). This approach seems adequate. It might be
40 possible to improve on it by recognizing that the risk at age "a" can be estimated by calculating
41 the 10-year risks at ages $a - 9$, $a - 8$, $a - 7$, etc. and averaging these. This alternative is not
42 recommended because it seems unlikely to affect the model results by much and does not seem
43 to justify the complication.

1 *iii. Please comment on the scientific validity of using the ASCVD risk model for*
2 *estimating reduced CVD risk stemming from changes in total cholesterol in*
3 *response to reducing exposure to PFOA and PFOS in drinking water.*
4

5 The ASCVD model is calibrated to data from epidemiological studies that establish a
6 relationship between total cholesterol and CVD risk. Such studies do not by themselves provide
7 evidence that a change in total cholesterol will change CVD risk, nor do they provide
8 information about whether the effect of a change in total cholesterol on CVD risk depends on the
9 source of the change. Intervention studies provide evidence about the joint effects of an
10 intervention (e.g., statins) on total cholesterol and CVD risk. However, such studies are unlikely
11 to provide evidence about the causal effect on CVD risk of a change in total cholesterol due to
12 reducing exposure to PFOA/PFOS in drinking water. It appears the validity of using the model in
13 this context can be assessed only by scientific judgment about the plausibility that a change in
14 total cholesterol due to a change in drinking water exposure has the same relationship to CVD
15 risk as a change due to sources that have been evaluated by intervention or observational studies.
16 Whether or not all components in the ASCVD risk model reach the threshold for POD derivation
17 (e.g., blood pressure) or statistical significance in a meta-analysis (e.g., HDLC), the decision not
18 to consider PFOA and PFOS effects on other parameters in the ASCVD model when estimating
19 avoided CVD risk as a result of the reduction of PFOA and PFOS in drinking water requires
20 further justification. This is of particular concern for HDL cholesterol which has been shown to
21 have similar discrimination for ASCVD risk in certain populations when included in the model.
22 The Panel recommends that EPA evaluate whether inclusion of HDL would influence the results
23 of the modeling.
24

25 **Recommendations**

26
27 The Panel recommends that EPA provide further discussion of the accuracy of the model
28 predictions in sub-groups with varying levels of social deprivation.
29

30 The Panel recommends that EPA evaluate whether inclusion of HDL would influence the results
31 of the modeling.
32
33

34 **Charge Question #4- Limitations and Uncertainties**

35 *Section 7 and Appendix A describe the limitations and uncertainties of the CVD risk reduction*
36 *analysis. Has EPA clearly described the individual contributions of the sources of uncertainty?*
37

38 Appendix E of the draft CVD document describes EPA's planned approach to uncertainty
39 analysis, which focuses on uncertainties in each step of the CVD risk reduction model
40 (PFOA/PFOS serum concentration inputs, PWS population size and demographics, impacts of
41 PFOA/PFOS on TC, risk of CVD event given TC and non-TC predictors, life tables, CVD
42 mortality and non-fatal events), most of which will be modeled directly using Monte Carlo
43 simulation. Section 7 of the EPA draft CVD document describes many sources of uncertainty
44 that will not be modeled directly and suggests the likely direction of the effect of each such
45 source on the CVD outcome estimates (i.e., underestimate, uncertain, or overestimate).

1
2 The Panel agreed that EPA was generally clear in describing the individual contribution of
3 sources of uncertainty in Section 7 and Appendix E of the Analysis of CVD Risk Reduction
4 document, and the approach to characterizing some uncertainties using Monte Carlo analysis in
5 Appendix E of the EPA draft CVD document. Additional specific areas needing further
6 clarification are listed in Appendix A of this report.
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8 **Recommendations**

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10 See Appendix A of this report for specific areas needing clarification.
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APPENDIX A

The Panel provided comments regarding specific areas requiring more thorough and clearer descriptions and revisions and additions for both the draft EPA main CVD document and CVD Appendix.

CVD risk reduction main draft document

- P. 11. EPA assumed drinking water accounts for 20% of total daily PFOA/PFOS dose under the baseline scenario, and that the other 80% is independent of drinking water PFOA/PFOS concentration. It tested the effect of alternatively assuming that drinking water accounts for 80% of the total and found “the assumption about drinking water source contribution does not affect the estimated changes in serum PFOA/PFOS.” The Panel suggests that EPA explain why the effect of a change in drinking-water PFOA/PFOS concentration on serum PFOA/PFOS should not be four times as large in the alternative case.
- P. 14. “EPA adjusts the modeled population cohort to exclude individuals with pre-existing conditions.” Row 1 in Table 3 cites national level CVD prevalence statistics, but the Panel finds that more detail is needed, as well as a reference to the Appendix (if applicable) for how the adjustment is done. Specifically, by how much does this adjustment reduce overall population numbers and which pre-existing conditions are excluded?
- P. 16. the Panel concludes that the EPA should report the calibration factors used in Table 3 and discuss how reasonable these magnitudes appear. Do these calibration factors inform how reasonable the projections from the ASCVD model are?
- P. 21. “ASCVD-based estimates may be inconsistent with the recent CVD prevalence statistics.” The Panel seeks clarification as to the level of inconsistency.

EPA Appendix B

- P. 28. How is integer age (a) defined? There are multiple statements like “at the beginning of integer age a and calendar year t .” Is integer age the age an individual reaches in the calendar year? Obviously, age and year do not change on the same date, unless everyone is born on New Year’s Day.
- P. 28. The definition of $l_{b,a,s,r,t,p}$ should presumably say “we assume that people who have just been born do NOT have CVD history by definition” (the word “not” is missing).
- P. 41. Table B-8 Post-Acute All-Cause Mortality After the First Myocardial Infarction shows all-cause death risk is lower for men than women who are non-Hispanic Blacks aged 45-64, but higher for men than women in all other age/race groups. Is this correct?
- P. 46. In “The uncalibrated number of persons experiencing their first hard CVD event in year t , $l_{b,a,s,r,t}$ ” should the $l_{b,a,s,r,t}$ be $\tilde{n}_{b,a,s,r,t}$?
- P. 46. “EPA applies a constant baseline annual probability of first hard CVD event estimated at age 80 to those currently aged 81–89 years.” What is known about baseline probability for ages 81-89? If the probability rises with age, this assumption will lead to an underestimate.

- 1 • P. 47. Should the first term in brackets in eqn. (B-12), $[1 + \sum_{f \in F} (\mu_{a,s,r,f,0} - 1) \cdot \gamma_{a,s,r,f}]$
- 2 be $\sum_{f \in F} (\mu_{a,s,r,f,0} \cdot \gamma_{a,s,r,f})$, i.e., the sum over types of nonfatal CVD events of the
- 3 product of the mortality rate and share by type? It would be useful to explain the logic of
- 4 the term as written. The same question applies to eqn. (B-23) on p. 48.
- 5 • P. 48. How is the term $\rho_{b,a,s,r}$ calculated? Is it a rate of post-acute CVD mortality?
- 6 • P. 49. Equations B-34 and B-36 may contain a sign error. As written, avoided deaths
- 7 $\Delta m_{b,a,s,r,t}$ and ΔM_t are both less than zero (when treatment is associated with lower
- 8 incidence of first CVD events, as stated).

9 Suggestions for revision

- 10 1. Quantified uncertainty about the slope of the relationships between TC and either PFOA
- 11 or PFOS should be clarified and should account for the sensitivity of the meta-analysis
- 12 results to restrictions on the functional form of the included estimates. It is not clear
- 13 whether the confidence intervals reported in Table E-1 are based on the variance of the
- 14 error term for individual observations or include some contribution for between-model
- 15 variation (e.g., model-specific random error τ^2). Moreover, uncertainty about the
- 16 functional form of the relationship seems likely to be important. The meta-regressions
- 17 use only linear models, but the central estimates using all studies or using only linear-log
- 18 models for TC both fall below the lower end of the confidence interval for the linear
- 19 models for PFOA (Table A-2), though not for PFOS (Table A-3).
- 20
- 21 2. Organization of Table 7 – it is difficult to tell from Table 7 and the discussion in Section
- 22 7 of the report which sources of uncharacterized uncertainty are likely to be most
- 23 significant in terms of the likely magnitude of their impact on the estimates. Some row
- 24 entries include a fairly detailed discussion (with sources cited), and others are much
- 25 sparser. Ideally, Table 7 would list the biggest concerns with respect to uncharacterized
- 26 uncertainty first, and offer the most detail for those, with smaller concerns listed later.
- 27
- 28 3. Requested specific clarifications
- 29 a. Why are systolic blood pressure and other CVD-related outcomes excluded? Row 7 in
- 30 Table 7 notes their exclusion, but does not explain it.
- 31 b. In row 10 of Table 7, why does non-linearity in the ASCVD model lead to
- 32 underestimation of CVD impacts when EPA uses the fraction of population that smokes
- 33 and has diabetes (compared to when the model uses binary values for an individual
- 34 patient)? In addition, a paper has recently been published in *JAMA Cardiology*, using the
- 35 Framingham cohort study data, showing that inclusion of former smoking status, pack-
- 36 years and years since quitting smoking improves ASCVD risk prediction among White
- 37 individuals over the reference model with 2013 PCE variables (Duncan et al,
- 38 doi:10.1001/jamacardio.2021.4990). The results require replication in other racial and
- 39 ethnic groups, but EPA may want to consider this approach to inclusion of detailed
- 40 smoking variables in their implementation of the ASCVD model.
- 41 c. We suggest adding some detail to row 4 of Table 7 to explain what is meant by “high-
- 42 quality data.” (We realize that the study selection for the meta-analysis is described in
- 43

- 1 detail in Appendix A, but a very brief list of selection criteria for “high-quality” studies
2 would be useful to include in Table 7.)
- 3 d. Is there empirical evidence supporting the assumption of independent effects of
4 PFOA/PFOS on TC, and if so, can sources be cited in row 6 of Table 7?
- 5
- 6 4. Requested specific additions
- 7 a. Row 7 of Table 7 notes that the exclusion of non-TC CVD-related outcomes results in an
8 “underestimate” of the CVD impacts of reducing PFOA/PFOS exposure. However, this is
9 not correct with respect to any impacts of PFOA/PFOS on HDLC which, as noted in the
10 discussion on pp. 11-13, is negatively associated with CVD events. Because EPA
11 included HDLC as one of only two predictors of CVD events in its meta-analyses, this is
12 an important omission. The exclusion of HDLC effects should be added as a separate row
13 in Table 7 (with “overestimate” entered in the “effect on estimate” column), or should be
14 listed in the existing row 7, changing “effect on estimate” to “uncertain,” and explaining
15 this in the “details” column.
- 16 b. Table E-1 in Appendix E notes that EPA is “currently evaluating including the PK model
17 in the uncertainty analysis.” Team members were supportive of this, because uncertainty
18 around baseline (pre-policy) PFOA/PFOS concentrations seems important to model.
19 However, column 3 of this row entry in Table E-1 (“will the source be analyzed”) says
20 “No.” We suggest that this piece of the uncertainty analysis be carried out, and that the
21 column 3 entry be switched to “Yes.”
- 22 c. We suggest that EPA add a row to Table 7 describing the likely impact of excluding the
23 population of individuals with pre-existing conditions.
- 24 d. We suggest that EPA add a row to Table 7 describing the likely impact of having
25 assumed that no PWS households will engage in averting behavior (e.g., bottled water
26 purchases) to reduce PFOA/PFOS exposure.
- 27 e. We suggest that EPA add a row to Table 7 describing the uncertainty introduced by
28 linking county FIPS codes with PWS locations (given the lack of a spatial match between
29 county borders and PWS service territories). This is discussed in Appendix E (Table E-
30 1), but it would be helpful to mention it in Table 7, as well.
- 31 f. Some reviewers felt that it would be helpful for EPA to describe cardiovascular impacts
32 that are monetized in other EPA analyses that are not included here. For instance, EPA
33 (2011), cited on page 15, monetizes cardiovascular hospitalizations. Excluding these
34 impacts would be a limitation of the current study.
- 35