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DRAFT SEWAGE SLUDGE RISK ASSESSMENT FOR PERFLUOROOCTANOIC ACID (PFOA) CASRN 335-67-1 AND PERFLUOROOCTANE SULFONIC ACID (PFOS) CASRN 1763-23-1

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Inter-agency reviewers only evaluated the sections of this document related to plant and livestock uptake (2.5, 2.9.3.4 & 2.9.3.5).

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EXECUTIVE SUMMARY

The purpose of this draft document is to characterize the potential human health and environmental risks associated with land application, surface disposal, and incineration of sewage sludge that contains perfluorooctanoic acid (PFOA) or perfluorooctane sulfonic acid (PFOS). The draft risk assessment focuses on those living on or near impacted sites or those that rely primarily on their products (*e.g.*, food crops, animal products, drinking water); the draft risk assessment does not model risks for the general public. This draft risk assessment will help inform whether PFOA or PFOS, based on our current understanding of their toxicity, persistence, concentration, mobility, or potential for exposure, may be present in sewage sludge in concentrations which may adversely affect public health or the environment (Clean Water Act section 405(d)(2)(A), 33 U.S.C. 1345(d)(2)(A)). The United States Environmental Protection Agency (EPA) uses the term "biosolids" to mean sewage sludge that has been treated to meet the Clean Water Act (CWA) requirements and is intended to be applied to land as a soil amendment or fertilizer. This draft risk assessment is not a regulation and is not EPA guidance.

All wastewater treatment plants (WWTPs) treating domestic sewage generate sewage sludge that needs to be managed either by disposal or reuse. Based on recent data received by the EPA from certain large publicly owned treatment works (POTWs) in the states where the EPA is the permitting authority, 3.76 million dry metric tons (DMT) of sewage sludge is generated each year.[1] There are several ways in which sewage sludge is disposed of in the U.S. In 2022, approximately 56% of sewage sludge generated by these POTWs was land applied, 24% was landfilled, 3% was disposed of in a sewage sludge monofill, 16% was incinerated, and 1% was disposed of using another method. Decisions about how to manage sewage sludge are influenced by site-specific factors, including local landfill capacity, access to sewage sludge incinerators (SSIs), demand for biosolids for use as an agricultural soil amendment, proximity to disposal/reuse mechanisms (*i.e.*, land suitable for application, monofills, landfills, incinerators), efforts to reduce methane releases by diverting organics from landfills, and other economic or feasibility considerations. In some states, POTWs primarily rely on one use or disposal method (for example, POTWs in Rhode Island and Connecticut primarily incinerate sewage sludge; POTWs in Nebraska and Colorado primarily rely on agricultural land application; POTWs in Louisiana and Kentucky primarily dispose of sewage sludge in landfills). Other states have roughly equal numbers of POTWs employing each use and disposal strategy (for example, Michigan and New Hampshire).[2]

PFOA and PFOS are two chemicals in a large class of synthetic chemicals called per- and polyfluoroalkyl substances (PFAS). PFAS have been manufactured and used by a broad range of industries since the 1940s, and there are estimated to be thousands of PFAS present in the global marketplace that are used in many consumer, commercial, and industrial products. PFOA and PFOS have been widely studied, and they were once high production volume chemicals within the PFAS chemical class. PFAS manufacturers voluntarily phased out domestic manufacturing of PFOS by 2002 and of PFOA by 2015, and the EPA restricted their uses by Significant New Use Rules (SNURs) issued under section 5(a)(2) of the Toxic Substances Control Act (TSCA), 15 U.S.C. 2604(a)(2). Although domestic manufacturing of PFOA and

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[1] See Biosolids Annual Reports from states where EPA is the Biosolids Program permitting authority covering 2022 submitted to the EPA's Office of Enforcement and Compliance, https://www.epa.gov/biosolids/basic-information-about-sewage-sludge-and-biosolids#statistics

[2] See summaries of state sewage sludge use and disposal data, https://www.biosolidsdata.org/state-summaries

PFOS have been phased out and their uses restricted, multiple activities still result in PFOA, PFOS, and their precursors being released to WWTPs.

PFOA and PFOS were prioritized for biosolids risk assessment for several reasons. First, they are difficult to degrade or treat in wastewater treatment plants because they are non-volatile, non-biodegradable, and sorb to solids. Second, both PFOA and PFOS bioaccumulate in humans, plants, fish, and livestock and are persistent in the environment. Finally, these chemicals are highly toxic to human beings; the EPA has classified both chemicals as *likely to be carcinogenic to humans,* and the available human epidemiological and animal toxicological evidence indicates that they adversely impact developmental, cardiac, hepatic, and immune systems depending on exposure conditions.[3,4,5]

There are recent, well-documented examples of significantly elevated PFOA and PFOS concentrations in U.S. sewage sludge contaminated by industrial sources to wastewater treatment plants. Statewide surveys of sewage sludge also find that PFOA and PFOS are consistently detected at wastewater treatment plants that do not receive wastewater from industrial users of the chemicals. This widespread occurrence in sewage sludge is likely due to the historic or ongoing presence of PFOA, PFOS, and their precursors in consumer, commercial, and industrial products. Following land application of sewage sludge contaminated with PFOA or PFOS, these chemicals have been detected in soils, groundwater, livestock, crops, surface water, and game. Limited or no data are available on environmental releases associated with sewage sludge monofills or sewage sludge incinerators. Though data are available regarding groundwater and leachate contamination with PFAS at landfills accepting mixed municipal solid wastes, it is not clear the portion of this contamination that could be attributed to sewage sludge disposal.

The goal of this risk assessment is to describe the potential human health and environmental risks associated with the use and disposal practices regulated under CWA Section 405(d) and regulation 40 C.F.R. Part 503, Standards for the Use or Disposal of Sewage Sludge: land application, surface disposal (*e.g.,* disposal in sewage sludge monofills), and incineration of sewage sludge that contains PFOA or PFOS. Not all the scenarios described in the draft risk assessment may be common practice. The draft risk assessment does not assess human health or environmental risks associated with disposal in municipal solid waste landfills, a common management practice for disposal of sewage sludge, because that practice is regulated under the Resource Conservation and Recovery Act (RCRA) and the regulation 40 CFR Part 258, Criteria for Municipal Solid Waste Landfills. For the incineration scenario, the draft assessment does not provide quantitative risk estimates given significant data gaps related to PFOA and PFOS destruction efficiency during incineration and potential exposure to products of incomplete combustion. The findings presented in this draft risk assessment are preliminary. The EPA expects to publish a final risk assessment after reviewing public comments and revising the risk assessment accordingly.

Prior to the writing of this draft refined risk assessment, the EPA performed a screening-level risk analysis for PFOA and PFOS in sewage sludge using a high-end deterministic exposure model for a farm

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[3] US EPA, Office of Water Final Human Health Toxicity Assessment for PFOA (2024). 815R24006 and US EPA Office of Water Final Human Health Toxicity Assessment for PFOS (2024). 815R24007.

[4] US EPA, Guidelines for Carcinogen Risk Assessment (2005). EPA/630/P-03/001B. https://www.epaov/risk/guidelines-carcinogen-risk-assessment.

[5] US EPA, ORD staff handbook for developing IRIS assessments (2022). (EPA 600/R-22/268). https://cfpub.epa.gov/ncea/iris_drafts/recordisplay.cfm?deid=356370

family living on a pasture or crop farm (*see* Appendix E). This screening approach assumed high starting concentrations of PFOA and PFOS in sewage sludge and high consumption rates for each exposure pathway. The high-end screening model resulted in elevated risk levels for every human exposure pathway (*e.g.*, drinking water; consumption of fish, milk, beef, vegetables). Given the risk indicated in the screening-level assessment, the EPA continued to a refined risk assessment. In this refined risk assessment, the EPA assessed risks under median (*i.e.,* central tendency, 50th percentile), rather than high-end exposure conditions, to better understand the potential scope and magnitude of risks under different sewage sludge use and disposal scenarios. To complete the central tendency deterministic modeling steps of the refined risk assessment, the EPA (1) assessed available fate and transport models to select the best available models for PFOA and PFOS, and (2) parameterized the models with inputs and exposure factors to reflect median U.S. conditions and consumption behaviors.

The draft risk assessment is scoped to model risks to human populations because available data indicate that humans are more sensitive to PFOA and PFOS exposures than aquatic or terrestrial wildlife or livestock. For the land application scenarios, the EPA modeled potential PFOA and PFOS exposures and estimated human health risks under three scenarios: (1) application to a farm with majority pasture-raised dairy cows, beef cattle, or chickens (pasture farm scenario), (2) application to a farm growing fruits or vegetables (crop farm scenario), and (3) application to reclaim damaged soils such as an overgrazed pasture (reclamation scenario). For the surface disposal scenario, the EPA modeled potential PFOA or PFOS exposures via groundwater to those living near a lined or unlined surface disposal site. Due to uncertainties around PFOA and PFOS destruction when sewage sludge is incinerated, the EPA did not quantitatively model the sewage sludge incineration scenarios for this draft risk assessment; instead, the EPA qualitatively described potential risks to communities living near a sewage sludge incinerator.

Based on the central tendency modeling results presented in the draft risk assessment, the EPA finds that draft risk estimates exceed the agency’s acceptable human health risk thresholds for some pasture farm, food crop farm, and reclamation scenarios when assuming that the land-applied sewage sludge contains 1 part per billion (ppb) of PFOA or PFOS. The EPA also finds that there may be human health risks associated with drinking contaminated groundwater sourced near a surface disposal site when sewage sludge containing 1 ppb of PFOA or sewage sludge containing 4 to 5 ppb of PFOS is disposed in an unlined or clay-lined surface disposal unit.

The presence and magnitude of human health risks from sewage sludge use and disposal to those living on or near impacted properties or primarily relying on their products is expected to vary across regions and among properties depending on the concentration of PFOA and PFOS in sewage sludge; the number of applications; the amount land applied; the climate, geology, and hydrology at the use or disposal site; agronomic practices; human behavioral patterns (*e.g.*, drinking water ingestion rates, consumption rate of impacted products); and many other site-specific factors. Not all farms or disposal sites where sewage sludge containing PFOA or PFOS have been used or disposed of are expected to pose a risk to human health. For example, human health risks are expected to be lower when sewage sludge is applied to areas with protected groundwater, sites that are distant from surface waters used for fishing or as a drinking water source, and when applied to certain crops, such as grain, fuel, or fiber crops. However, the EPA’s modeling results from the draft risk assessment suggest that under certain scenarios and conditions, land-applying or disposing of sewage sludge containing a detectable level (*i.e.,* 1 ppb or more) of PFOA or PFOS could result in human health risks exceeding the agency’s acceptable thresholds for cancer and non-cancer effects.

Modeling for land application scenarios suggests that, when the majority of the consumer’s dietary intake of a product comes from a property impacted by the land application of sewage sludge, the highest risk pathways include (1) drinking milk from majority pasture-raised cows consuming contaminated forage, soil, and water, (2) drinking water sourced from contaminated surface or groundwater on or adjacent to the impacted property, (3) eating fish from a lake impacted by runoff from the impacted property, and (4) eating beef or eggs from majority pasture-raised hens or cattle where the pasture has received impacted sewage sludge. The risk calculations assume each of these farm products (*e.g.,* milk, beef, eggs) or drinking water consumed comes from the impacted property but does not combine risks from each of these products. The EPA did not estimate risk associated with occasionally consuming products impacted by land application of contaminated sewage sludge nor foods that come from a variety of sources (*e.g.*, milk from a grocery store that is sourced from many farms and mixed together before being bottled).

Draft risk estimates are presented in the risk assessment as cancer risk levels and hazard quotients (HQs). Cancer risk levels represent the number of expected excess lifetime cancer cases due to exposure to the carcinogenic pollutant in a given population size (*e.g.,* a cancer risk level of 1 in 1,000 indicates that lifetime exposure to the carcinogenic pollutant would be expected to cause one additional case of cancer for every one thousand people in the exposed population). Risk for noncancer effects are expressed as HQs that represent the ratio of the potential exposure to a pollutant to the level below which adverse noncancer effects are not expected (*i.e.,* an HQ of less than 1 means adverse noncancer health effects are unlikely and thus risk can be considered negligible; an HQ greater than 1 means adverse noncancer effects are possible and thus risk is indicated).

Risk estimates for the highest risk pathways can exceed the EPA's acceptable thresholds by several orders of magnitude. For example, for the land application scenarios, cancer risk levels associated with drinking the modeled amount of contaminated milk (*i.e.,* 32 ounces per day for adults) can exceed 1 in 1,000, and HQs for non-cancer effects associated with eating the modeled amount of contaminated fish (*i.e.,* 1 to 2 servings per week for adults) can reach up to 45. For the crop farm scenario, there are limited scientific studies available regarding the uptake of PFOA and PFOS from sewage sludge-amended soils into certain fruits and vegetables; however, the draft risk assessment suggests that cancer risks from consuming the modeled amount of these contaminated foods (*e.g.,* 1 serving per day for adults for certain categories of fruits and vegetables) can exceed 1 in 100,000 for PFOA. Because the draft risk assessment indicates risks associated with individual exposure pathways, there may be potential risks to populations beyond the farm family (*e.g.*, people living near a use or disposal site who use contaminated groundwater as a source of drinking water or people who primarily consume produce, dairy, or meat from a farm that has applied contaminated sewage sludge under the modeled conditions).

For the surface disposal sites, there are no exceedances of the EPA's risk thresholds for PFOA or PFOS in down-gradient groundwater at composite-lined surface disposal sites. However, for unlined and clay-lined surface disposal sites, there can be exceedances of the risk thresholds for the drinking water pathway: for unlined sites, the cancer risk levels can exceed 1 in 1,000 and HQs are as high as 12; for clay-lined sites, the cancer risk levels can exceed 1 in 1,000 and HQs are up to 9. As mentioned above, the draft risk assessment does not include quantitative risk estimates for incineration due to data limitations.

The draft risk calculations are not conservative estimates because they (1) model risks associated with sludge containing 1 ppb of PFOA or PFOS, which is on the low end of measured U.S. sewage sludge concentrations, (2) reflect median exposure conditions (*e.g.*, 50th percentile drinking water intake rates), (3) do not include non-sewage sludge exposures to PFOA or PFOS (*e.g.*, consumer products, other dietary sources), (4) do not account for the combined risk of PFOA and PFOS together, and (5) do not account for exposures from the transformation of PFOA or PFOS precursors. As such, risk estimates that account for multiple dietary exposures (*e.g.*, consuming impacted milk, water, and eggs), multiple sources of exposure (*e.g.*, exposure to PFOA or PFOS-containing consumer products), or exposure to other PFAS would be greater than those presented in this draft risk assessment. Further, the EPA's draft risk assessment relies on models where risks scale linearly with the starting concentration of PFOA or PFOS in sewage sludge. As such, sewage sludge containing ten times more PFOA or PFOS (*i.e.*, 10 ppb) would yield risk estimates that are ten times greater than those presented in the draft risk assessment (assuming all other factors are constant).

The EPA did not complete Monte Carlo probabilistic modeling because risks exceeding acceptable thresholds were identified in multiple scenarios and pathways in the central tendency deterministic modeling results. For example, in the EPA's draft risk assessment, when calculating risks from egg consumption in the central tendency approach, the model assumes that an adult living on a farm consumes, on average, 1 egg per day from the impacted property for ten years, which represents the median egg consumption rate for households who farm.[6] The model further assumes that when the adult lives on the impacted farm, they have no sources of PFOA or PFOS exposure other than eggs and that for the remainder of the adult's life, they have no exposure to PFOA or PFOS through any pathway. Since risk is indicated under this central tendency scenario, Monte Carlo probabilistic modeling, which would examine the entire distribution of potential exposures to PFOA or PFOS and report the 95th percentile of the risk distribution, is not warranted. For this reason, the EPA is not conducting additional modeling exercises at this time, but rather is focusing on sharing the central tendency modeling results and identifying actions that could be taken to mitigate risks.

In summary, the results of the draft risk assessment indicate that there are potential risks to human health to those living on or near impacted properties or primarily relying on their products from land application and surface disposal of sewage sludge containing PFOA and PFOS and that risk is dependent on (1) the concentration of PFOA and PFOS in sewage sludge, (2) the specific type of management practice (*e.g.*, type of land application or presence of a liner in a monofill), and (3) the local environmental and geological conditions (*e.g.*, climate and distance to groundwater). Risks are possible, though not quantified, from the incineration of PFOA and PFOS-containing sewage sludge. Site-specific factors should be considered when planning risk mitigation and management practices to reduce human exposures associated with PFOA and PFOS in sewage sludge.

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[6] See EPA's Exposure Factors Handbook, https://www.epa.gov/expobox/about-exposure-factors-handbook, Table 13-40

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LIST OF ACRONYMS

Acronym	**Definition**
1-D	1-dimensional
3-D	3-dimensional
3MRA	Multimedia, Multipathway, and Multireceptor Risk Assessment modeling system
ADD	average daily dose
AFFF	aqueous film forming foam
ALT	alanine transaminase
ATSDR	Agency for Toxic Substances and Disease Registry
AWI	air-water interface
AWQC	Ambient Water Quality Criteria
BAF	bioaccumulation factor
BAR	Biosolids Annual Report
BCF	bioconcentration factor
BST	Biosolids Tool
BTF	biotransfer factor
BW	body weight
CAA	Clean Air Act
CASRN	Chemical Abstracts Service registry number
CDC	Centers for Disease Control and Prevention
CDPHE	Colorado Department of Public Health & Environment
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CFR	Code of Federal Regulations
Class A_{EQ}	class A exceptional quality (biosolids)
COR	carry over rate
CR	consumption rate
CRL	cancer risk level
CSA	community supported agriculture
CSF	cancer slope factor
CT DEEP	Connecticut Department of Energy and Environmental Protection
CWA	Clean Water Act
diPAP	polyfluoroalkyl phosphate diesters
DMT	dry metric tons
DOC	dissolved organic carbon
DW	dry weight
EC_{25}	25% effect concentration
ECCC	Environment and Climate Change Canada
ED	exposure duration
EFH	Exposure Factors Handbook
EFSA	European Food Safety Authority
EPA	US Environmental Protection Agency
EPACMTP	EPA's Composite Model for Leachate Migration with Transformation Products
EXAMS	Exposure Analysis Modeling System
FDA	US Food and Drug Administration
FEQG	Federal Environmental Quality Guideline (Canada)
FGD	Flue gas desulfurization
f_{oc}	fraction of organic carbon
FOSAA	perfluorooctane sulfonamidoacetic acid
FOSE	perfluorooctane sulfonamidoethanol
FR	Federal Register
FTCA	fluorotelomer carboxylic acid
FTOH	fluorotelomer alcohol
FTP	fluorotelomer-based polymer
FTS	fluorotelomer sulfonate
GIS	geographic information system
GSAF	grass soil accumulation factor
GSCM	Generic Soil Column Model
GW	groundwater

Acronym	**Definition**
HELP	Hydrologic Evaluation of Landfill Performance model
HGDB	Hydrogeologic Database
HHRAP	Human Health Risk Assessment Protocol
HLC	Henry's law constant
HQ	hazard quotient
HSDB	Hazardous Substances Data Bank
HUC	hydrologic unit code
IC_{25}	25% inhibition concentration
IQR	interquartile range
IUR	inhalation unit risk
K_d	solid-phase adsorption coefficient
K_{oc}	organic carbon distribution coefficient
K_{ow}	water-octanol partitional coefficient
LADD	lifetime average daily dose
LAU	land application unit
LC	liquid chromatography
LC_{50}	50% lethal concentration
LD_{50}	50% lethal dose
LOD	limit of detection
LOQ	limit of quantification
LWS	Local Watershed Model
MAF	moisture adjustment factor
Maine DEP	Maine Department of Environmental Protection
MDL	method detection limit
MI EGLE	Michigan Department of Environment, Great Lakes, and Energy
MPART	Michigan PFAS Action Response Team
MRL	maximum residue level
MS	mass spectrometry
MSW	municipal solid waste
MW	molecular weight
n.d.	non-detect
NEtFOSA	N-ethyl perfluorooctane sulfonamide
NEtFOSAA	N-ethyl perfluorooctane sulfonamidoacetic acid
NEtFOSE	N-ethyl perfluorooctane sulfonamidoethanol
NFCS	Nationwide Food Consumption Survey
NH DES	New Hampshire Department of Environmental Services
NHANES	National Health and Nutrition Examination Survey
NIST	National Institute of Standards and Technology
NLM	National Library of Medicine
NMeFOSAA	N-methylperfluorooctane sulfonamidoacetic acid
NSSS	National Sewage Sludge Survey
OC	organic carbon
OM	organic matter
OPP	Office of Pesticide Programs
PAN	plant available nitrogen
PAP	polyfluoroalkyl phosphoric acid
PBPK	physiologically based pharmacokinetic
PCBs	polychlorinated biphenyls
PEM	Particulate Emissions Model
PFAA	polyfluoralkyl acids
PFAS	per- and polyfluoroalkyl substances
PFBS	perfluorobutane sulfonic acid
PFCAs	perfluoroalkyl carboxylic acids
PFDA	Perfluorodecanoic acid
PFDoDA	Perfluorododecanoic acid
PFHpA	Perfluoroheptanoic acid
PFHxA	perfluorohexanoic acid
PFHxS	perfluorohexane sulfonic acid

Acronym	**Definition**
PFHxS	perfluorohexane sulfonic acid
PFI	polyfluorinated iodide
PFNA	Perfluorononanoic acid
PFOA	perfluorooctanoic acid
PFOS	perfluorooctane sulfonic acid
PFOSA	perfluorooctanesulfonamide
PFSA	perfluorosulfonic acid
PFUnDA	Perfluoroundecanoic acid
PIC	product of incomplete combustion
pKa	acid dissociation constant
POM	percent organic matter
POTW	publicly owned treatment works
ppb	parts per billion
ppt	parts per trillion
RAGS	Risk Assessment Guidance for Superfund
RCF	root concentration factor
RCRA	Resource Conservation and Recovery Act
RfC	reference concentration
RfD	reference dose
RSC	relative source contribution
RSL	regional screening level
sAmPAP	perfluorooctane sulfonamidoethanol-based phosphate diester
SAMSON	Solar and Meteorological Surface Observation Network
SATK	saturated hydraulic conductivity
SCS	Soil Conservation Service
SDU	surface disposal unit
SFEI	San Francisco Estuary Institute
SI	surface impoundment
SNUR	Significant New Use Rule
SPM	suspended particulate matter
SSI	sewage sludge incinerators
SSURGO	Soil Survey Geographic database
STATSGO	State Soil Geographic database
SW	surface water
TNSSS	Targeted National Sewage Sludge Survey Sampling and Analysis Technical Report
TOP	total oxidizable precursors (assay)
TRI	Toxics Release Inventory
TSCA	Toxic Substances Control Act
TSDF	treatment, storage, and disposal facility
TSS	total suspended solids
USDA	U.S. Department of Agriculture
USGS	U.S. Geological Survey
USLE	Universal Soil Loss Equation
VT DEC	Vermont Department of Environmental Conservation
VVWM	Variable Volume Waterbody Model
WBAN	Weather Bureau-Army-Navy
WW	wet weight
WWTP	wastewater treatment plant

1 BACKGROUND

1.1 Clean Water Act Section 405 Authority

Section 405(d) of the Clean Water Act (CWA), 33 U.S.C. 1345(d), requires the United States Environmental Protection Agency (EPA) to establish numerical limitations and management practices, when appropriate, that protect public health and the environment from the reasonably anticipated adverse effects of toxic pollutants in sewage sludge. Section 405(d) also requires the EPA to review sewage sludge regulations at least every two years for the purpose of identifying additional pollutants that may be present in sewage sludge and, if appropriate, to propose practices and standards for those pollutants consistent with the requirements set forth in the CWA.

Section 405(e) of the CWA, 33 U.S.C. 1345(e), prohibits any person from disposing of sewage sludge from a publicly owned treatment works (POTW) or other treatment works treating domestic sewage through any use or disposal practice for which regulations have been established pursuant to Section 405 except in compliance with the Section 405 regulations at 40 CFR part 503. Section 405(g) of the CWA, 33 U.S.C. 1345(g), authorizes the EPA to conduct public information projects and to disseminate information pertaining to the safe use of sewage sludge.

In 1993, the EPA promulgated final regulations regarding sewage sludge, the “Standards for the Use or Disposal of Sewage Sludge” (40 CFR Part 503). That rule contains management practices and pollutant limits that protect public health and the environment from reasonably anticipated adverse effects of ten regulated pollutants in sewage sludge when the sewage sludge is land applied, placed in a surface disposal unit, or fired in a sewage sludge incinerator. The terms “biosolids” and “sewage sludge” are often used interchangeably by the public; however, the EPA typically uses the term “biosolids” to mean sewage sludge that has been treated to meet the requirements in Part 503 and is intended to be applied to land as a soil amendment or fertilizer. The EPA’s rules and the CWA only use the term “sewage sludge.”

1.2 Purpose

The goal of human health and ecological risk assessment is to estimate the nature and probability of adverse health effects in humans or other ecological populations that may be exposed to chemicals in contaminated environmental media, now or in the future. The risk assessment process includes 1) planning the scope of the assessment, 2) identifying the hazards by describing how the stressor has the potential to cause harm to humans and/or ecological systems, 3) assessing exposures to the humans and ecological receptors and 4) characterizing the risks to those exposed human and ecological populations. Risk assessments also include a discussion of areas of uncertainty and variability in the assessment.

Perfluorooctanoic acid (PFOA) and perfluorosulfonic acid (PFOS) are two chemicals within the family of fluorinated organic substances called per- and polyfluoroalkyl substances (PFAS). The purpose of this draft risk assessment is to assess the potential human health and environmental risks associated with land application and disposal of sewage sludge that contains PFOA or PFOS. This draft risk assessment considers several common use and disposal scenarios for sewage sludge and the resulting exposures to aquatic and terrestrial wildlife, in addition to impacted human populations. There are four detailed sewage sludge modeling scenarios described in this document: reuse (land application) on a farm growing fruits and vegetables (*crop farm scenario*), reuse (land application) on a farm raising livestock (*pasture farm scenario*), disposal in a surface disposal site (*surface disposal scenario*), and reuse (land application) to restore degraded soils (*land reclamation scenario*). Potentially impacted human populations included in the modeled scenarios are farm families, those drinking water impacted by sewage sludge disposal sites or biosolids land application sites, participants in community supported agriculture[7] (CSA), those growing food in home or community gardens, and those who eat freshwater fish.

In this document, Section 2 (Problem Formulation) describes the scope of the draft risk assessment and assessment endpoints for PFOA and PFOS. Section 3 (Analysis) presents estimated concentrations in relevant media for exposure, such as groundwater and soil. Section 4 (Risk Characterization) includes risk estimation and risk description. Section 5 (Uncertainty, Variability, and Sensitivity) describes how uncertainty may affect the draft risk assessment. Finally, Section 6 (Comparison of Modeled Concentrations and Observed Concentrations in Relevant Media) compares modeled results from this draft assessment to biosolids investigations in various locations.

This draft risk assessment is not a regulation and is not EPA guidance. Furthermore, the draft risk assessment does not include a discussion of risk management options. The draft risk assessment was externally peer reviewed through a task order with a contractor.[8] A panel of five scientists reviewed the draft risk assessment and responded to charge questions through the contractor on August 6, 2024. The peer reviewers' comments and the EPA's responses are available in a separate document (US EPA, 2024n).

1.3 Use and Disposal of Sewage Sludge

Each year, certain large POTWs[9] in the United States are required to summarize their sewage sludge management practices and compile compliance information in Biosolids Annual Reports (BARs). The EPA collects BARs from roughly 2,500 facilities in the 41 states where the EPA is the permitting authority.[10] These POTWs generate approximately 3.76 million dry metric tons (DMT) of sewage sludge each year that either needs to be disposed of or reused. Disposal options include landfilling, incineration, and other disposal methods like deep well injection. Landfilling can occur in a sewage sludge monofill (*i.e.*, surface disposal) which is regulated under the CWA in 40 CFR Part 503, but most landfilling occurs at municipal solid waste (MSW) landfills, which are regulated under RCRA in 40 CFR part 258 and will not be part of this assessment. Based on the BARs covering 2022 from facilities where the EPA is the permitting authority, approximately 27% of all generated sewage sludge was landfilled, 16% was incinerated, and 1% was disposed of using another method. Reuse of sewage sludge is often preferred by treatment works because it tends to be less costly, produces fewer carbon emissions, and/or provides a benefit as a soil amendment. Reuse options include land application on agricultural lands, at reclamation sites, or at home gardens or other sites like golf courses, often through the sale of bulk or bagged product. As of 2022, land application at agricultural sites accounted for 31% of sewage

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[7] Community Supported Agriculture is an arrangement where consumers purchase a share of produce typically from one or a small number of farmers. Commonly a variety of produce will be included in the arrangement so that purchasers receive regular deliveries throughout the local growing season.

[8] Versar under Contract No. 68HERH23A0021 Task Order 68HERH23F0320

[9] BARs are required from by POTWs that 1) serve 10,000 people or more; 2) are Major POTWs (POTWs with a design flow rate greater than or equal to one million gallons per day); 3) are Class 1 Management Facilities (POTWs with an approved pretreatment program or facilities that have been classified as such by the EPA or state); or are otherwise required to report by EPA or permitting authority, that land apply, surface dispose or incinerate in a sewage sludge incinerator. The EPA does not receive data from smaller POTWs, private or federal treatment works, or those that use alternate use or disposal practices like landfilling except on a voluntary basis.

[10] There are nine states (Arizona, Idaho, Michigan, Ohio, Oklahoma, South Dakota, Texas, Utah, and Wisconsin) that are authorized through the National Pollutant Discharge Elimination System (NPDES) Program to be the permitting authority for biosolids. The EPA will transition to electronic reporting for the remaining authorized states as part of Phase 2 implementation of the NPDES eRule by December 2025.

sludge use, land application at reclamation sites accounted for 1%, and other land application accounted for 24% (Figure 1). Overall, about 56% of all sewage sludge generated is land applied.

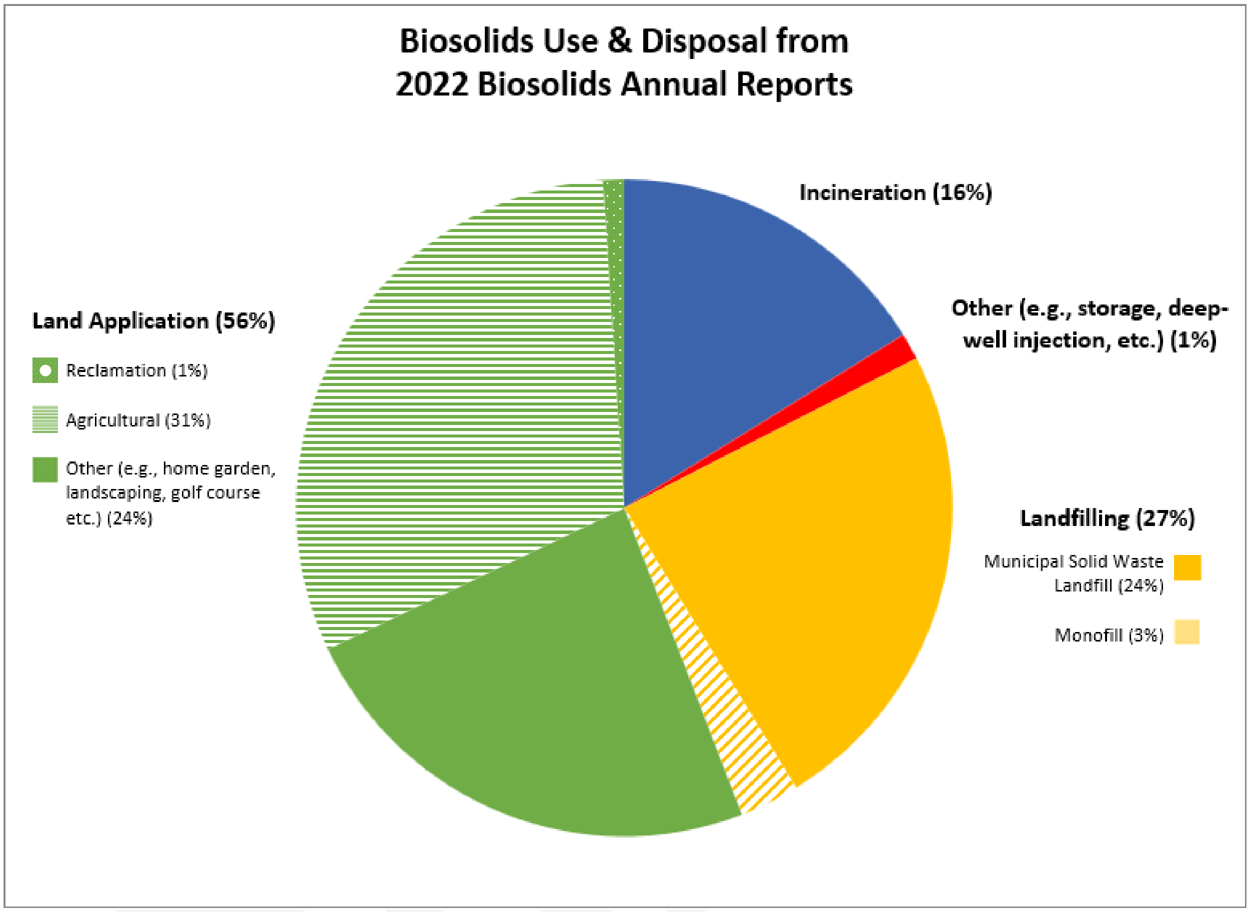


Figure 1. Distribution of sewage sludge use and disposal from Biosolids Annual Reports covering 2022 submitted to the EPA's Office of Enforcement and Compliance.

Current regulations for the land application of sewage sludge at 40 CFR part 503 require land application at the agronomic rate for nitrogen. 40 CFR § 503.14(d). The main exception is when the goal of land application is reclamation of a site that has been degraded (*e.g.*, repairing the surface of a mining site); in such cases, sewage sludge can be applied above the agronomic rate to restore organic material and encourage vegetative regrowth. 40 CFR § 503.14(d). Biosolids land application can also be conducted as frequently as desired if the agronomic, pathogen, and vector attraction requirements within 40 CFR part 503 are met for the crop or farming activity (note that domestic septage, which is defined as the liquid or solid material removed from septic tanks, cesspools, portable toilets, Type III marine sanitary devices, or similar systems, can be similarly land applied at application rates which are based on agronomic rates for nitrogen. 40 CFR § 503.13(c). Additionally, home gardeners who purchase or receive bulk or bagged biosolids are not required to apply biosolids at an agronomic rate. 40 CFR §§ 503.10(b)-(g).

Surface disposal is the placement of sewage sludge onto land for final disposal in a sewage sludge unit (*e.g.*, sewage sludge-only landfill or "monofill"). 40 CFR § 503.21(n). Requirements for surface disposal in Part 503 include placement restrictions, methane monitoring, and pollutant limits where applicable, among others. Surface disposal sites may be unlined or lined with leachate collection systems. Preamble to 40 CFR § 503, 58 FR 9301, February 19, 1993. There are no chemical pollutant limits in Part 503 for surface disposal sites with a liner and leachate collection system. Unlined surface disposal sites must meet the applicable pollutant requirements in Part 503, 40 CFR §§ 503.23(a)-(b). Liners at surface disposal sites would be required if the sewage sludge exceeds contamination levels for certain metals in 40 CFR part 503, 40 CFR §§ 503.23(a)-(b). The only restrictions on distance to adjacent properties from surface disposal are based on the contamination levels of the sewage sludge with arsenic, chromium, and nickel at an unlined surface disposal site. 40 CFR § 503.23(a)(2).

Sewage sludge incinerators (SSIs) are regulated by Part 503 under the CWA and under the Clean Air Act (CAA). Requirements for incineration in Part 503 include stack monitoring and pollutant concentrations, among others. 40 CFR § 503 Subpart E. Pollutant limits in sewage sludge fed into an incinerator are based on risk specific concentrations calculated using dispersion factors and operating parameters including stack height. 40 CFR § 503.43. Sewage sludge incineration regulations allow higher dispersion factors for stack heights over 65 meters. 40 CFR § 503.43. The EPA's rules regarding emissions from SSIs were updated in 2016. More information on the EPA's CAA regulations for SSIs can be found on EPA's website for the New Source Performance Standards and Emission Guidelines (US EPA, 2023a).

1.4 History of Sewage Sludge Risk Assessment

In 1987, the US Congress passed the Water Quality Act, which amended the CWA to require the EPA to establish a comprehensive program to reduce potential environmental risks associated with sewage sludge management and maximize the beneficial reuse of sewage sludge. As amended, Section 405(d) of the CWA required the EPA to establish numerical limits and management practices that protect public health and the environment from the reasonably anticipated adverse effects of toxic pollutants in sewage sludge. The amendment required two rounds of sewage sludge regulations and set deadlines for the EPA to establish those regulations. In 1993, the EPA promulgated the first rule (called "Round One," 40 CFR part 503), which set numeric limits in sewage sludge for ten metals (arsenic, cadmium, chromium, copper, lead, mercury, molybdenum, nickel, selenium, and zinc). In that action, the EPA further identified 31 pollutants and pollutant categories to be prioritized for the second planned rule (called "Round Two"). On October 25, 1995 (60 FR 54763), chromium land application pollutant limits were withdrawn, the selenium limits were modified, and the EPA narrowed the original list of 31 prioritized pollutants to two pollutant groups for the second round of rulemaking: polychlorinated dibenzo-p-dioxins/dibenzofurans and dioxin-like co-planar polychlorinated biphenyls (PCBs) (US EPA 1996). On December 23, 1999, the EPA proposed numeric limits for dioxins, dibenzofurans, and co-planar PCBs (also called "dioxin-like PCBs") in sewage sludge applied to land and proposed not to regulate dioxins in sewage sludge disposed of in a surface disposal unit or fired in a sewage sludge incinerator (64 FR 72045). On June 12, 2002, the EPA published a Notice of Data Availability containing new information related to dioxins in land-applied sewage sludge and requested public comments (67 FR 40554). Based on these new data and revised risk assessment conclusions, on October 24, 2003, the EPA determined that regulation of dioxins in sewage sludge was not warranted (68 FR 61084). The supporting technical documentation for the 1993 "Round One" regulation and the 2003 "Round Two" determination not to regulate put forward a general framework for sewage sludge risk assessment that is used for this draft risk assessment of PFOA and PFOS.

As described above, the EPA's previous sewage sludge risk assessments have assessed uses and disposal options for sewage sludge that potentially present risk to humans, crops, livestock, or wildlife (US EPA, 1992; US EPA, 1995a; US EPA, 2003a). In the 1992 technical support document, the EPA based numerical limits for sewage sludge when applied to agricultural land on a modeled assessment of the potential risk to public health and the environment through 14 pathways of exposure related to land application or disposal. These pathways were split into two categories: pathways relevant to agricultural land and pathways relevant to non-agricultural land. Agricultural land application scenarios included use of sewage sludge by a farmer for food or feed crops on pasture or rangeland, including large farms or home gardeners. Non-agricultural use and disposal scenarios included use on forest land; land reclamation sites; "public contact sites," which may include lands like golf courses; and surface disposal sites. When evaluating risks associated with sewage sludge that is incinerated, the EPA assessed a single pathway of exposure – inhalation – and did not include air transport and deposition onto soils or surface waters. The 14 pathways of exposure modeled in the 1992 assessment were as follows; the exposed individual for each pathway with a human receptor is listed in square brackets:

- Reuse (land application)
  - Sludge-soil-plant-human [consumer] (pathway 1)
  - Sludge-soil-plant-human [home gardener] (pathway 2)
  - Sludge-soil-human [child] (pathway 3)
  - Sludge-soil-plant-animal-human [farm household] (pathway 4)
  - Sludge-soil-animal - human [farm household] (pathway 5)
  - Sludge soil-plant-animal (pathway 6)
  - Sludge-soil-animal (pathway 7)
  - Sludge-soil-plant (pathway 8)
  - Sludge-soil- soil organism (pathway 9)
  - Sludge-soil-soil organism-soil organism predator (pathway 10)
  - Sludge-soil-airborne dusts-human [tractor operator] (pathway 11)
  - Sludge-soil-surface water -human [person consuming drinking water and fish] (pathway 12)
  - Sludge-soil-air-human [off-site resident] (pathway 13)
  - Sludge-soil-groundwater-human [person consuming drinking water] (pathway 14)
- Surface disposal
  - Sludge-soil-air-human [off-site resident] (pathway 13)
  - Sludge soil groundwater-human [person consuming drinking water] (pathway 14)
- Incineration
  - Sludge-incineration particulate -air-human [off-site resident] (pathway 13).

A graphical depiction of each of the pathways evaluated in this risk assessment is presented in Section 2.8.

As described in the 1992 technical support document, the farm family was considered to be the most exposed population to land applied sewage sludge due to their potential exposures to consuming their own crops and interacting directly with the contaminated soils. All the human-health based regulations were protective of the incidental soil ingestion pathway for children because this pathway was considered to be sensitive for human health across all life stages and potential exposure pathways. Chemicals were also assessed for ecological risk including risk to crop growth and livestock that fed on those plants.

In the second round of risk assessment, the EPA considered dioxin-like compounds to be the only chemicals that merited a full risk assessment. The EPA performed a Monte Carlo analysis of exposure to the farm family using national sewage sludge survey concentrations to estimate exposures across the dietary pathways established in the 1993 regulations, with minor adjustments to allow for the assessment of specific animal products (such as milk) relevant to dioxins. The risk assessment aggregated ingestion exposures pathways (milk, meat, soil etc.) and included a cumulative assessment across chemicals in the dioxin category (US EPA, 2003a;b). The EPA later concluded that the 95th percentile exposures from this assessment did not exceed the hazard-based reference doses in the assessment (US EPA, 2003c). This conclusion justified the decision to not regulate PCBs or dioxins in any use or disposal practice for sewage sludge based on the risk levels estimated for highly exposed populations (US EPA, 2003c).

This assessment for PFOA and PFOS follows the general frameworks set out in the EPA's 1992 and 2003 assessments, with some modifications to account for the chemical and environmental characteristics of PFOA and PFOS.

2 PROBLEM FORMULATION

2.1 Literature Search Strategy and Information Management

Risk assessment for land application and disposal of sewage sludge requires the synthesis of available information from a diverse set of academic fields of research: chemical occurrence in sewage sludge, environmental fate and transport, human toxicology, aquatic toxicology, plant toxicity, and wildlife or ecological effects. The assessment further benefits from a background understanding of the chemical's use profile in the U.S. economy and the uses or disposal options common for sewage sludge generated in the U.S. To efficiently synthesize this information, the EPA takes a hierarchical approach to information management. When possible, the EPA sources background information and risk assessment conclusions from publicly available, peer-reviewed documents such as EPA Human Health Toxicity Assessments, Health Effects Support Documents, Aquatic Life Ambient Water Quality Criteria, Agency for Toxic Substances and Disease Registry (ATSDR) Toxicological Profiles, Environment and Climate Change Canada (ECCC) Federal Environmental Quality Guidelines, European Food Safety Authority (EFSA) Scientific Opinions, and other such assessments. When assessments are not available on a specific topic or not up to date with current scientific findings, the EPA conducts literature reviews of peer-reviewed journal articles and state agency "gray literature" reports. Background information summarized in the Problem Formulation (Section 2) of this assessment is based on existing assessments. The literature search strategies employed for the model parameters are described in Model Parameterization (Section 2.9.3).

2.2 The Nature of the Chemical Stressor

2.2.1 Chemical Identity

PFOA and PFOS are manufactured for direct use in industry and in commerce, in addition to a range of other chemical structures containing fluorinated carbons (Buck et al., 2011; OECD, 2021; US EPA, 2021b). Some of these other PFAS can degrade in the environment to PFOA or PFOS, which are then stable degradation and metabolic products. The PFAS that degrade to PFOA and PFOS are called precursors. Generally, precursors to PFOA and PFOS also contain a fluorinated carbon chain with eight or more carbons.

PFOA and PFOS have been part of a voluntary phase out for domestic manufacture and their uses have been restricted by Significant New Use Rules (SNURs) issued by the EPA under the Toxic Substances Control Act (TSCA) and the US Food and Drug Administration (FDA) phase out for food packaging (FDA, 2016; US EPA, 2024a). While these actions may have reduced the presence of these chemicals in domestic sewage, PFOA and PFOS continue to be detected in wastewater and sewage across the U.S. due to their presence in residential, commercial, and industrial products that were manufactured or imported before the phase-out, their presence in products or processes associated with the limited number of ongoing allowable uses (US EPA, 2021c), their persistence in waste disposal sites like landfills, and their pervasive existing environmental contamination (see Sections 2.2 and 2.3).

PFOA: PFOA is a synthetic fluorinated organic chemical that has been manufactured and used in a variety of industries since the 1940s (US EPA, 2018a). The chemical abstracts service registry number (CASRN), common synonyms, chemical formula, and other basic chemical properties are described in Table 1. PFOA repels water and oil, is chemically and thermally stable, and exhibits surfactant properties. Based on these properties, it has been used in the manufacture of many materials, including cosmetics, paints, polishes, and nonstick coatings on fabrics, paper, and cookware. It is very persistent in the human body and the environment (Calafat et al., 2007; 2019). More information about PFOA's uses and properties can be found in the EPA's 2024 Final Human Health Toxicity Assessments for PFOA (US EPA, 2024b). In 2006, the EPA invited eight major companies to commit to working toward the elimination of their production and use of PFOA (and chemicals that degrade to PFOA) and elimination of these chemicals from emissions and products by the end of 2015. All eight companies have since phased out manufacturing PFOA. Despite this commitment of these major producers, PFOA may be produced, imported, and used by companies not participating in the PFOA Stewardship Program and some uses of PFOA are ongoing (see 40 CFR 721.9582). PFOA is included in EPA's SNUR issued in January 2015, which ensures that the EPA will have an opportunity to review any efforts to reintroduce the chemical into the marketplace and take action, as necessary, to address potential concerns (US EPA, 2015). Limited existing uses of PFOA-related chemicals, including as a component of anti-reflective coatings in the production of semiconductors, were excluded from the regulations (US EPA, 2021c) and PFOA may still be a component of articles (manufactured items) imported into the U.S.

Table 1. Chemical and Physical Properties of PFOA.

Property	PFOA, acidic form[1]	Source
CASRN	335-67-1	NA
Chemical Abstracts Index Name	2,2,3,3,4,4,5,5,6,6,7,7,8,8,8-pentadecafluorooctanoic acid	NA
Synonyms	PFOA; Pentadecafluoro-1-octanoic acid; Pentadecafluoro-n-octanoic acid; Octanoic acid, pentadecafluoro-; Perfluorocaprylic acid; Pentadecafluorooctanoic acid; Perfluoroheptanecarboxylic acid;	NA
Chemical Formula	$C_8HF_{15}O_2$	NA
Molecular Weight (grams per mole [g/mol])	414.07	PubChem Identifier (CID 9554) (URL: https://pubchem.ncbi.nlm.nih.gov/compound/9554); Lide (2007)
Color/Physical State	White powder (ammonium salt)	PubChem Identifier (CID 9554) (URL: https://pubchem.ncbi.nlm.nih.gov/compound/9554)
Boiling Point	192.4 ºC	Lide (2007); SRC (2016)
Melting Point	54.3 ºC	Lide (2007); SRC (2016)
Vapor Pressure	0.525 mm Hg at 25 °C (measured) 0.962 mm Hg at 59.25 °C (measured)	Hekster et al. (2003); SRC (2016) ATSDR (2021); Kaiser et al. (2005)
K_{AW}	0.00102 (experimentally determined, equivalent to Henry's Law Constant of 0.000028 Pa-m^3/mol at 25 °C)	Li et al. (2007)
K_{OW}	Not measurable	UNEP (2015)
pKa	3.15 (mean measured)	Burns et al. (2008) and 3M (2003) as reported in EPA Chemistry Dashboard (URL: https://comptox.epa.gov/dashboard/dsstoxdb/results?search=DTXSID8031865#properties)
Solubility in Water	9,500 mg/L (estimated); 3,300 mg/L at 25 ºC (measured)	Hekster et al. (2003); ATSDR (2021)

[1] PFOA is most commonly produced as an ammonium salt (CASRN 3825-26-1). Properties specific to the salt are not included.

PFOS: PFOS is a synthetic fluorinated organic chemical that has been manufactured and used in a variety of industries since the 1940s (US EPA, 2018a). The CASRN, common synonyms, chemical formula, and other basic chemical properties are described in Table 2. Like PFOA, PFOS repels water and oil, is chemically and thermally stable, and exhibits surfactant properties. Based on these properties, it has been used in the manufacture of many materials, including cosmetics, paints, polishes, and nonstick coatings on fabrics, paper, and cookware. Like PFOA, PFOS is very persistent in the human body and the environment (Calafat et al., 2007; 2019). More information about PFOS's uses and properties can be found in the EPA's 2024 Final Human Health Toxicity Assessments for PFOS (US EPA, 2024c). In 2000, the principal manufacturer of PFOS agreed to a voluntary phase-out of PFOS production and use over time. This phase-out agreement was completed in 2002 (US EPA, 2007). PFOS is included in EPA's SNUR issued in December 2002, which ensures that the EPA will have an opportunity to review any efforts to reintroduce PFOS into the marketplace and take action, as necessary, to address potential concerns (US EPA, 2002). Limited existing uses of PFOS-related chemicals, including as an anti-erosion additive in fire-resistant aviation hydraulic fluids and as a component of antireflective coating in the production of semiconductors, were excluded from the regulation (US EPA, 2013) and articles imported into the U.S. may have PFOS. Due to the high human health toxicity of PFOS, all environmental releases may be significant; however, known major sources of PFOS contamination in the U.S. include past manufacturing of PFOS, use of PFOS as a mist suppressant in chrome plating facilities, use of PFOS as an oil and water-resistant coating for paper products, textiles, and leather, and use of PFOS-containing firefighting foams, especially at training and testing sites.

Table 2. Chemical and Physical Properties of PFOS

Property	PFOS, acidic form[1]	Source
CASRN	1763-23-1	NA
Chemical Abstracts Index Name	1,1,2,2,3,3,4,4,5,5,6,6,7,7,8,8,8- heptadecafluoro-1-octanesulfonic acid	NA
Synonyms	Perfluorooctane sulfonic acid; heptadecafluoro-1-octane sulfonic acid; PFOS acid; perfluorooctane sulfonate	NA
Chemical Formula	$C_8HF_{17}O_3S$	NA
Molecular Weight (grams per mole [g/mol])	500.13	Lewis (ed. 2004); SRC (2016)
Color/Physical State	White powder (potassium salt)	OECD (2002)
Boiling Point	258–260 °C	SRC (2016)
Melting Point	No data	
Vapor Pressure	2.48 x 10^{-6} mm Hg at 20°C (potassium salt)	ATSDR (2021)
Henry's Law Constant	Not measurable; not expected to volatilize from aqueous solution (< 2.0 x 10^{-6})	ATSDR (2021)
K_{OW}	Not measurable	EFSA (2008); ATSDR (2021)
pKa (modeled)	0.14 (no empirical measurements available)	ATSDR (2021)
Solubility in Water	570-680 mg/L	OECD (2002); ATSDR (2021)

[1] PFOS is commonly produced as a potassium salt (CASRN 2795-39-3). Properties specific to the salt are not included.

Tables 1 and 2 include a summary of physical and chemical properties for PFOA and PFOS. While these values provide important context for understanding the general behaviors of the chemical, when assessing the relevance of reported physical properties to their behavior in the soil environment, it is important to ensure that the method for collecting the physical and chemical data is relevant to the environmental conditions modeled in the risk assessment. For example, measurements of volatility like vapor pressure or Henry's law constant performed on the acid at low pH (Li, 2007) may be useful for understanding PFOA or PFOS in a laboratory or industrial setting, but farm fields tend to have pH values closer to neutral pH where PFOA and PFOS exist as an anion. Using these physical property values directly to estimate volatility from a farm field may be misleading. Section 2.9.3 of this document describes the physical and chemical properties used to parameterize models used in this risk assessment and describes how studies were selected to best capture relevant environmental conditions.

2.2.2 Transformation and Degradation of Precursors

PFOA and PFOS do not undergo degradation under environmentally relevant conditions (US EPA, 2008; OECD, 2002; Schultz et al., 2003), in part because environmental degradation pathways and processes do not apply enough energy to break fluorine-carbon bonds (3M, 2000; Hekster et al., 2003; Schultz et al., 2003). ATSDR Toxicological Profiles for Perfluoroalkyls (including PFOA and PFOS) conclude that these perfluoroalkyl acids are resistant to biodegradation, direct photolysis, atmospheric photooxidation, and hydrolysis (ATSDR, 2021; OECD, 2002; Prevedouros et al., 2006). Some researchers are exploring the potential for degradation in soil systems that are undergoing remediation (Huang et al., 2022).

The processing of influent and sewage sludge at wastewater treatment plants (WWTPs) provides opportunities for fluorinated precursors to biodegrade to PFOA or PFOS, which are terminal degradants. Examples of precursors to PFOS include perfluorooctane sulfonamidoethanol-based phosphate diester (sAmPAPs) containing carbon-chain moieties with at least eight fluorinated carbons, *N*-ethyl perfluorooctane sulfonamidoethanol (NEtFOSE), *N*-ethyl perfluorooctane sulfonamidoacetic acid (NEtFOSAA), perfluorooctane sulfonamidoethanol (FOSE), perfluorooctane sulfonamidoacetic acid (FOSAA), *N*-ethyl perfluorooctane sulfonamide (NEtFOSA), and perfluorooctanesulfonamide (PFOSA). Commonly detected precursors to PFOA include fluorotelomer alcohols (FTOHs), polyfluoroalkyl phosphoric acids (PAPs) and polyfluorinated iodides (PFIs) that contain a fluorinated carbon chain moiety with at least eight carbons in the chain (*i.e.*, 8:2 FTOH). Sidechain fluorotelomer-based polymers (FTPs), especially those used on textiles, could also be significant sources of PFOA to WWTPs because they can transform when laundered or cleaned and when weathered in soils (Washington et al., 2015; Washington & Jenkins, 2015; Liagkouridis et al., 2022; van der Veen et al., 2022). The treatment of sewage sludge to create biosolids (Thompson et al., 2023a) and the land application of biosolids (Schaefer et al., 2022) both provide opportunities for precursors to degrade into PFOA and PFOS.

PFOA and PFOS precursors have been used by industry and imported in consumer products. When these chemicals enter the environment, the molar yields for their transformation to PFOA or PFOS and their degradation rates vary. Laboratory measurements have shown that microbes common to WWTPs (Lange, 2000) and other environmental systems can biodegrade these precursors to PFOA and PFOS. Biosolids-amended soil in column studies have observed that the degradation of PFAS precursors may be responsible for a significant portion of PFOA and PFOS that occur in the environment (Schaefer et al., 2022).

Due to data gaps regarding the occurrence, environmental fate and transport, degradation pathways, bioaccumulation, and toxicity of precursors to PFOA and PFOS, the EPA is focusing this draft risk assessment on PFOA and PFOS. That said, the occurrence data of PFAS in biosolids indicate precursors significantly contribute to the overall PFOA and PFOS loading to soils and disposal facilities (see Section 2.4). Future assessments could be expanded to include other chemicals including environmental precursors to PFOA and PFOS, or other PFAS. Additionally, policy decisions regarding the treatment of quantifiable precursors to PFOA and PFOS could be considered in the future.

2.2.3 Environmental Fate and Transport

PFOA and PFOS are persistent in the environment and are commonly called “forever chemicals” due to the lack of observed degradation pathways. They are also mobile in the environment and bioaccumulate in organisms. The EPA and state monitoring programs have found that historic land application of sewage sludge containing PFOA and PFOS has contaminated soil, surface water, groundwater, crops, beef, eggs, and milk and impacted farm families (Washington et al., 2010; Lindstrom et al., 2011; Yoo et al., 2011; Moavenzadeh Ghaznavi et al., 2023).

PFOA and PFOS can undergo several transport mechanisms after release to a soil environment. These include sorption to soils and sediments, sorption to fluid-fluid interphases, runoff, erosion, migration to groundwater, and uptake into plants and animals. The surfactant-like properties of PFOA and PFOS influence the way they move through natural systems. For example, because PFOA and PFOS sorb to fluid-fluid interfaces (Sharifan et al., 2021) some modeling assumptions used for other organic chemicals are not appropriate for PFOA and PFOS. PFOA and PFOS exhibit varying partitioning between soil and water, air and water, or biosolids and water depending on the presence and type of organic matter (Ebrahimi et al., 2021), oxalate-extractable grain coatings, mineral composition (Gravesen et al., 2023), the presence of air-water interfaces (Costanza et al., 2019) and other factors (Sharifan et al., 2021). These properties have been shown in the literature to result in a wide range of potential values of soil-water and air-water sorption constants across different types of soils (see Appendix C). The degree of soil-water and air-water sorption influences transport behavior from the soil to other media like groundwater and surface water.

Several studies characterize PFAS partitioning behavior between the solid and aqueous phases in sewage sludges (Zhang et al., 2013; Ebrahimi et al., 2021; Lewis et al., 2023; Gravesen et al., 2023). While a correlation has been found between bulk organic matter content and PFAS partitioning (particularly for long-chain PFAS), research has shown that protein content has the strongest correlation to PFAS partitioning in biosolids when compared to lipids and bulk organic matter, which aligns with the observation that PFOA and PFOS bind to proteins in animals (Zhang et al., 2013; Ebrahimi et al., 2021, Section 2.5.1). Also, a more recent study investigated the effects of microbial weathering on PFAS partitioning over time after biosolids land application to examine the fate and transport of PFAS leaching from biosolids into the environment (Lewis et al., 2023). Results revealed that microbial weathering plays a role in PFAS partitioning, contributing to the biodegradation of organic matter and leading to an increased potential for PFAS leaching to groundwater. Another recent study examined oxalate-extractable iron and aluminum in relation to PFAS partitioning in biosolids, finding that iron was correlated with PFOA partitioning and aluminum was correlated with both PFOA and PFOS partitioning (Gravesen et al., 2023). In addition, bulk organic matter was associated with PFOS partitioning, while protein content tended to be more strongly correlated with the partitioning of shorter-chain PFAS (Gravesen et al., 2023).

The partitioning trends described in the prior paragraph are observed in sewage sludge and are also relevant to organic-matter rich topsoils that have been amended with biosolids; PFOA and PFOS partitioning behavior in subsurface soils is distinct due to lower organic content, differences in the mineral or amorphous mineraloid composition of grains and grain coatings, and the presence of air-water interphases. Due to the low concentrations of natural organic matter in subsurface soils (0.01-0.05%), PFAS sorption in the subsurface may have significant contributions from sorption to the surfaces of minerals and mineraloids and sorption to the air-water interphases (Lyu et al., 2019). Most studies in this area have been lab-based tests in well-defined media such as quartz sand or limestone, which differ from natural soil systems. Additional study is needed on the most significant variables related to PFOA and PFOS retention in natural subsurface soil systems.

Although volatilization of PFOA and PFOS is expected to be low from soil systems in general due to the chemicals being ionized at typical soil pH, there may be soil systems where volatilization contributes to atmospheric concentrations. Past research regarding soil-water environments has shown that PFAS volatilization increases as pH decreases (Johansson et al., 2017; Sima and Jaffé, 2021). In an experiment examining water-air transfer, highest rates of PFOA volatilization occurred at a pH of 1, while PFOA volatilization was found to be negligible at pH levels greater than 2.5 (Johansson et al., 2017). Consequently, under natural soil-water conditions, PFOA volatilization is considered to be negligible (Johansson et al., 2017; Sima and Jaffé, 2021). However, under natural soil-water conditions, there could be a concern with the volatilization of precursors that can biodegrade and transform into persistent PFAS. For example, in a past study, as much as 3% of 6:2 polyfluoroalkyl phosphate diester (diPAP, a precursor that can transform into perfluorohexanoic acid [PFHxA], for instance) volatilized under natural soil conditions, while 8:2 diPAP (a precursor of PFOA, for example) was found to be negligible (Liu and Liu, 2016). More study is needed on the volatilization of other PFOA or PFOS precursors under natural soil conditions.

2.3 Sources to Wastewater Treatment Plants and Biosolids

The EPA Chemical Data Reporting rule under TSCA requires manufacturers (including importers) to report certain data about chemicals in commerce in the U.S., including information on PFOA and PFOS (subject to a 2,500 pound reporting threshold at a single site). The last time PFOA and PFOS manufacturing information was reported to the EPA pursuant to this rule was in 2013 and 2002, respectively. However, Toxics Release Inventory (TRI) data for 2020 shows that PFOA and PFOS continue to be released into the environment. Pursuant to TRI reporting requirements, facilities in regulated industry sectors must report annually on releases and other waste management of certain listed toxic chemicals that they manufacture, process, or otherwise use above certain threshold quantities (currently 100 pounds per industrial site for PFOA and PFOS).

Despite the phase out of domestic manufacturing of PFOA and PFOS, multiple activities result in PFOA, PFOS, and their precursors being present in WWTP influent including industrial releases (*e.g.*, semiconductor manufacturing, pulp and paper plants), commercial releases (*e.g.*, hotels, car washes, industrial launderers), and down the drain releases from homes (*e.g.*, laundering of coated textiles, use of residential products). These chemicals have been used in a variety of industrial processes and commercial and consumer products, which results in a range of potential sources to WWTPs within communities. For example, homes may still have PFOA and PFOS-containing products in use, like after-market water resistant sprays, floor finishes, textiles with PFOA and PFOS coatings, or ski wax. These products could be washed down a drain or released when cleaned or laundered, or they may be disposed of at a lined MSW landfill. The leachate from that landfill could be another ongoing source of PFOA and PFOS to WWTPs, as the most common off-site management practice for landfill leachate is transfer to a WWTP (US EPA, 2024g). At different WWTPs across the country, any of these release mechanisms may play a role in PFAS entering the plant.

Sewage sludge contaminant monitoring based on typical analytical methods (*e.g.*, EPA Method 1633, US EPA, 2024d) can be used to test for 40 PFAS but does not include precursors such as sAmPAPs and diPAPs. Several studies using soil columns and non-targeted analysis have found that most of the environmental loading to biosolids will come from these precursor chemicals (Schaefer et al., 2022; Thompson et al., 2023a;b).

2.4 Occurrence in Biosolids

Studies have shown that PFAS are frequently found in biosolids around the globe (D’eon et al., 2009; Yoo et al., 2009; Lee et al., 2010; Washington et al., 2010; Lindstrom et al., 2011; Sepulvado et al., 2011; Venkatesan and Halden, 2013; Lee et al., 2014; Armstrong et al., 2016; Navarro et al., 2016; Eriksson et al., 2017; Moodie et al., 2021; Munoz et al., 2022; Fredriksson et al., 2022; Helmer et al., 2022; Johnson, 2022; Schaefer et al., 2022; Thompson et al., 2023a,b; Link et al., 2024). For a summary of PFOA and PFOS concentrations found in biosolids in the U.S. based on studies from published peer-reviewed literature and state reports, please see Appendix A, Tables A-1 (PFOA) and A-2 (PFOS). Overall, these studies have demonstrated that PFOS is typically found more often and at higher concentrations than PFOA in biosolids. Concentrations are reported on a dry weight basis. This review focuses on concentrations that occur as most studies do not identify sources of PFOA or PFOS.

The PFOA and PFOS concentrations found in U.S. biosolids vary across studies (Appendix A). At the national scale, the Venkatesan and Halden 2013 study measured 13 PFAS in composite samples compiled from archived biosolids collected during the EPA's 2001 National Sewage Sludge Survey (NSSS). The study authors randomly divided the 110 available samples from the 2001 NSSS (94 POTWs) into 5 composite samples, finding average concentrations of 34±22 parts per billion (ppb) for PFOA and 403±127 ppb for PFOS (Venkatesan and Halden, 2013). PFOA concentrations ranged from 12-70 ppb and PFOS concentrations ranged from 308-618 ppb (Venkatesan and Halden, 2013). Of the 13 PFAS analytes measured, 10 were detected in all composited samples with PFOS found at the highest levels, surpassing PFOA, which had the second highest concentrations (Venkatesan and Halden, 2013). A more recent U.S. study found, on average, lower concentrations of PFOA and PFOS in biosolids: PFOA concentrations ranged from 0.8-8.12 ppb and PFOS concentrations ranged from 0.386-150 ppb in samples collected from multiple states (7 WWTPs with a variety of treatment processes in urban areas receiving both industrial and domestic sources) (Schaefer et al., 2022). Another recent U.S. study analyzed samples before and after treatment from 8 WWTPs representing the four most common biosolids treatment processes in Florida, finding PFOA concentrations ranging from 1.7-21 ppb (before treatment) and 1.1-7.7 ppb (after treatment), and PFOS concentrations ranging from 4-41 ppb (before treatment) and 1.4-19 ppb (after treatment) (Thompson et al., 2023a). Though these studies with samples collected after the PFOA and PFOS phased out in the U.S. observe lower levels of PFOA and PFOS in sewage sludge than pre-phase out samples, sewage sludge samples with significantly elevated concentrations of PFOA and PFOS have been identified from industrially impacted WWTPs as recently as 2022 (Link et al., 2024).

Several states also have implemented programs to monitor for PFAS in their biosolids. For instance, Michigan's extensive sampling found that PFAS levels tended to be higher in biosolids receiving industrial sources (MI EGLE, 2021a,b; 2022; Helmer et al., 2022). Consequently, Michigan instigated industrial pretreatment program best management practices to limit PFAS source contributions (MI EGLE, 2021a, 2022; Helmer et al., 2022), which most recently has led to a PFOS reduction in biosolids of more than 85% at four of the six wastewater treatment plants studied (MI EGLE, 2022). A recent study analyzed Michigan's statewide biosolids data collected between 2018 and 2022 from 190 wastewater treatment plants representing both industrial and domestic sources, finding mean dry weight concentrations of 4.8±11 ppb for PFOA with a detection rate of 63% and 40±179 ppb for PFOS with a detection rate of 95% (Link et al., 2024). Based on biosolids data in Maine's Environmental and Geographic Analysis Database collected from 2019 to 2022, Maine's comprehensive state sampling found mean PFOA concentrations of 9.4 ppb in 2019, 8.2 ppb in 2020, 5.3 ppb in 2021, and 6.6 ppb in 2022, and mean PFOS concentrations of 27.2 ppb in 2019, 16.6 ppb in 2020, 22.7 ppb in 2021, and 19.3 ppb in 2022 (Brown and Caldwell, 2023). New Hampshire also has performed detailed PFAS analyses of soils, biosolids, solid/water partitioning, and groundwater leaching through a three-phase study conducted by the US Geological Survey and the New Hampshire Department of Environmental Services (Phase 1: Santangelo et al., 2022; Phase 2: Tokranov et al., 2023; Phase 3: Santangelo et al., 2023). Data from the finished biosolids collected from facilities in 2021 as part of Phase 2 found PFOA and PFOS dry weight concentrations of less than 8 ppb across samples (Tokranov et al., 2023).

The EPA is currently planning for the next NSSS in collaboration with the Effluent Guidelines Program's POTW Influent Study, both of which will focus on testing for PFAS ([March 26, 2024] (89 FR 20962); [October 10, 2024] (89 FR 82238)). This joint monitoring study will provide a current and comprehensive national dataset of PFAS concentrations found in the influent, effluent, and sewage sludge of POTWs, as well as their industrial and domestic sources.

As previously described, precursors also can transform into PFOA or PFOS in primary and secondary processes of wastewater treatment plants and in the environment. As an example, diPAPs can biodegrade and transform into persistent perfluoroalkyl carboxylic acids (PFCAs) (*e.g.*, 8:2 diPAP can transform into PFOA) (D'eon and Mabury, 2007; Lee et al., 2010; 2014), potentially leading to increased PFCA loads in wastewater effluent and land-applied biosolids that can contribute to aquatic and terrestrial contamination (Lee et al., 2010; 2014). A recent field study in Germany demonstrated that diPAPs stemming from paper production have the capacity to transform into PFCAs that can leach out of soil into drinking water sources (Lämmer et al., 2022). The Schaefer et al. 2022 study not only tested U.S. WWTP biosolids, but also performed column mesocosm leaching experiments, finding that precursors to the 18 measured polyfluoroalkyl acids (PFAAs; *e.g.*, diPAPs, 5:3 fluorotelomer carboxylic acid [FTCA], perfluorophosphonic acids) accounted for over 75% of the total PFAS fluorine mass in biosolids (Schaefer et al., 2022). Notably, this study found that total oxidizable precursor assay (TOP assay) in biosolids extracts was not able to quantify all precursors to PFOA and PFOS because the assay did not fully oxidize precursors like diPAPs. In addition, the Thompson et al. 2023a study analyzed 92 PFAS analytes in total, including precursors, and found that 6:2 diPAP, 6:2/8:2 diPAP, and 8:2 diPAP were the most common diPAPs identified in biosolids (Thompson et al., 2023a). The study results also showed that there is currently an underestimation of total PFAS concentrations due to the high potential for precursor transformation and lack of available analytical test methods that include these precursors in their targeted list of PFAS analytes (Thompson et al., 2023a). In another recent article analyzing toilet paper samples from the U.S. and other countries, along with U.S. sludge samples, 6:2 diPAP was detected at the highest concentrations in both toilet paper and sludge samples (Thompson et al., 2023b). Though some of these precursors do not transform to PFOA and PFOS, conducting non-targeted analysis and including more precursors in targeted methods can aid in resolving this issue of identifying unknown PFOA and PFOS precursors. Appendix A (Table A-3) provides examples of occurrence data for potential PFOA and PFOS precursors found in biosolids in the U.S. based on recent studies.

Despite the phase-out of long-chain PFAS (*e.g.*, PFOA and PFOS), the most recent U.S. studies still show that PFOS is typically found at the highest concentrations in biosolids of the traditional targeted list of PFAS analytes measured (Helmer et al., 2022; Link et al., 2024). Recent investigations in Michigan that include industrially impacted biosolids have shown PFOS concentrations as high as 2,150 ppb (MI EGLE, 2022; Link et al., 2024) and 6,500 ppb (MI EGLE, 2021a; Helmer et al., 2022); Michigan implemented industrial pretreatment program best management practices to address these PFOS sources and these concentrations have been reduced (MI EGLE, 2021a; 2022; Helmer et al., 2022). Michigan did not include PFOA and PFOS precursors in their industrial pretreatment and biosolids management strategy. Schaefer et al. 2022 found that concentrations of 8:2 diPAP exceeded concentrations of PFOA in biosolids. Schaefer et al. 2023 found that the sum of N-ethyl perfluorooctanesulfonamide (NEtFOSA), PFOSA, 8:2 fluorotelomer sulfonate (8:2 FTS), 8:2 FTCA, and N-methyl perfluorooctanesulfonamide acetic acid (NMeFOSAA) exceeded concentrations of PFOA and PFOS in biosolids. Thompson et al. 2023a found that the sum of 8:2 diPAP, 6:2/8:2 diPAP, FOSAA, NMeFOSAA, NEtFOSAA and 8:2 FTS concentrations also exceeded PFOA and PFOS concentrations in biosolids, with 8:2 diPAP and 6:2/8:2 diPAP being the most significant contributors to the total measured PFOA and PFOS precursor concentration. 8:2 diPAP and 6:2/8:2 diPAP are not currently included in EPA's analytical method recommended for sewage sludge, EPA 1633 (US EPA, 2024d).

2.5 Uptake and Bioaccumulation

This section provides a brief overview of PFOA and PFOS accumulation into animals (Section 2.5.1) and plants (Section 2.5.2). There are several characteristics of PFOA and PFOS uptake in humans, other animals, and plants that are important to understanding the overall fate and toxicity of these chemicals in biosolids-specific environmental release scenarios. A detailed description of how each uptake factor is parameterized in biosolids fate and transport models can be found in Section 2.9.3. Overall, PFOA and PFOS bioaccumulate in humans, fish, livestock, wildlife, and plants.

2.5.1 Animals

Humans: PFOA and PFOS accumulate in humans, and a detailed description of human absorption, distribution, metabolism, and elimination for PFOA and PFOS is available in the EPA's Final Human Health Toxicity Assessments (US EPA, 2024b;c). In contrast to many persistent organic pollutants that tend to partition to fats, PFOA and PFOS preferentially bind to proteins (Martin et al., 2003a;b). Within the body, PFOA and PFOS tend to bioaccumulate within protein-rich tissues, such as the blood serum proteins, liver, kidney, and gall bladder (De Silva et al., 2009; Martin et al., 2003a;b). Half-lives in humans differ by sex due to the elimination pathway of menstruation, lactation, and childbirth for women. PFOA and PFOS undergo enterohepatic recirculation, in which PFOA and PFOS are excreted from the liver in bile to the small intestine, then reabsorbed and transported back to the liver (GoeckeFlora and Reo, 1996). Reuptake also occurs through the kidneys (US EPA, 2024b;c). This reabsorption is one reason why PFOA and PFOS are retained for long time periods in the human body, and in the bodies of some other animals. PFOA and PFOS can be passed from mother to child *in utero* (through placental transfer) and in early life through breastmilk (US EPA, 2024b;c).

The Centers for Disease Control and Prevention (CDC) National Health and Nutrition Examination Survey (NHANES) has measured blood serum concentrations of several PFAS in the general U.S. population since 1999. PFOA and PFOS have consistently been detected in up to 98% of serum samples collected in biomonitoring studies that are representative of the U.S. general population. However, blood levels of PFOA declined by more than 60% between 1999 and 2014, presumably due to restrictions on PFOA commercial usage in the US (CDC, 2017). Blood levels of PFOS similarly declined by more than 80% between 1999 and 2014, a decline which also coincides with restrictions on PFOS commercial usage in the U.S. (CDC, 2017). Serum levels of people living in regions impacted by point source releases of PFOA, PFOS, and their precursors have elevated serum levels compared to the general population (MDH, 2010; US EPA, 2024b;c). For example, a 2024 study in Maine of 30 individuals from 19 households who have been living on farms with PFAS contamination for an average of 23.7 years found that this group's serum levels of PFOA, PFOS and other PFAS were significantly higher than the general population (Criswell et al., 2024). Further, the authors found that the farm families had serum levels similar to those seen in other highly exposed populations, including the C8 study cohort (Criswell et al., 2024; Frisbee et al., 2009).

Other animals: PFOA and PFOS are consistently detected in aquatic and terrestrial animals across the globe (Giesy and Kannan, 2001; US EPA, 2024l,m; De Silva et al., 2021). Accumulation is observed in game species (deer, ducks, fish) as well as other wildlife (Death et al., 2021). In wildlife, PFOS is generally observed with a higher frequency of detection and concentration than PFOA. In several areas with point sources of PFOS to the environment, state agencies have issued consumption restriction advisories for fish and game (MDHHS, 2023; MDIFW, 2021; MPCA, 2023a; NCDHHS, 2023).

Just as there are sex differences in the elimination rate of PFOA and PFOS in humans, these sex differences have also been observed in non-human animal species. For example, Lee and Schultz 2010 observed that the elimination rate of PFOA from blood plasma was ten times faster in female fathead minnows compared to males. The faster elimination rate may be related to sex hormone (*i.e.*, androgen and estrogen) levels, as the elimination rate in females decreased four-fold following exposure to the androgen trenbolone (Lee and Schultz, 2010). This pattern has also been observed in rats, where the elimination of PFOA was 70 times faster in females than males, which was attributed to sex-related differences in the expression of organic anion transporters in kidneys (Kudo et al., 2002). The degree to which sex-related differences in elimination rate apply to other fish species, or other taxonomic groups, may vary.

The EPA recently published a summary of PFAS occurrence information in freshwater fish from randomly selected sampling points in the U.S. portion of the Great Lakes (US EPA, 2024h). This study finds that PFOS is detected in the edible filets of 100% of freshwater fish samples, while PFOA was detected in 23% of samples. The range of PFOS concentrations found in filets is 0.366 to 49.3 ng/g. The range of detected concentrations of PFOA in fish filets is 0.086 to 1.41 ng/g. Both dissolved PFOS and sediment-sorbed PFOS contribute to the uptake of PFOS into freshwater fish (Balgooyen & Remucal, 2022; Barbo et al., 2023).

2.5.2 Plants

Uptake of PFOA and PFOS from biosolids-amended soils by crops is a potential pathway for entry into the human food chain, and plant uptake generally is one of several potential pathways for wildlife and livestock exposures. Generally, the degree of phytoaccumulation of a given chemical from soil to plants is either assessed with greenhouse-based lab studies or field-based studies. Previous biosolids risk assessments for metals indicated that greenhouse studies tended to result in higher measured uptake from soil to plants than field studies (US EPA, 1992). This assessment hypothesized that the differences in observed uptake of metals could be due to 1) increased transpiration in humid greenhouses, 2) higher concentrations of soluble salts in greenhouse pot soil due to the application of nutrients in a limited soil volume, which increases diffusion of metals from soil particles to roots, 3) soil acidification in greenhouse pots due to application of certain fertilizers in a small soil volume, which results in increased metal uptake, and 4) the soil-sludge mixture in greenhouse pots comprise the entire rooting medium, while in the field, sludge amended soils only extend to the tillage depth, and roots can extend below this depth. Some of these factors are also applicable to PFOA and PFOS uptake in plants, and consideration is needed of these factors when assessing plant uptake studies.

It has long been known that PFOA and PFOS can accumulate in plants eaten by humans (D'Hollander et al., 2015; 3M, 2001). Few studies have measured plant uptake data available for biosolids amended soil at field sites, but some PFOA and PFOS data from these biosolids-specific field studies are available (Yoo et al., 2011; Blaine et al., 2013). Both of these field-based studies are useful for understanding uptake into forage and silage, which improves the strength of the assessment of exposures to pastured livestock due to diet. Data on PFOA and PFOS accumulation into other plant species (*e.g.*, human food crops like fruits and vegetables) grown in biosolids-amended fields are somewhat limited; this represents a data gap for biosolids risk assessment because these measurements are the most direct way to understand exposures to humans who consume crops (fruits and vegetables) from biosolids-amended soils.

In October 2023, the EPA announced a new funding opportunity for research that furthers our understanding of PFAS uptake and bioaccumulation in plants and animals in agricultural, rural, and Tribal communities (US EPA, 2023b). The EPA is also aware of several ongoing studies regarding PFAS and plant uptake from biosolids-amended soils, which will likely be completed by the end of 2024. Ideally, as more data are collected in this area and the mechanistic understanding of PFOA and PFOS uptake into various types of plants and plant compartments improves, the uncertainty regarding exposure modeling from plants to humans, livestock, and wildlife will decrease.

PFOA and PFOS are taken up into various compartments of plants. There are many types of study design that measure PFOA and PFOS plant uptake, including laboratory studies that grew plants in natural soils, spiked soils, or spiked water, and field studies investigating plants grown in soil that have or have not been amended with biosolids. Some studies focus on accumulation in plants consumed by humans or animal feed (Yoo et al., 2011; Lechner and Knapp, 2011; Lee et al., 2014; Blaine et al., 2014; Bizkarguenaga et al., 2016a,b; Wen et al., 2016; Liu et al., 2017; Navarro et al., 2016; Ghisi et al., 2019; Kim et al., 2019; Li et al., 2019), others focus on how accumulation intersects with phytotoxicity (Lin et al., 2020; Zhou et al., 2016), and finally others look for "hyperaccumulating" plants including aquatic plants like pondweed and water-starwort, or terrestrial plants like long beech fern, sunflower, and hemp (Li et al., 2021; Nassazzi et al., 2023).

PFOA and PFOS both accumulate in food and feed crops grown in biosolid-amended soils. These studies generally indicate that uptake is stronger into the vegetative parts of plants (stems, leaves) than the edible part of the plant (fruit, seeds). For example, PFOA and PFOS accumulation factors are higher in corn silage than in corn grain (Simones et al., 2023). It is hypothesized that PFOA and PFOS accumulation is higher in shoot or root crops due to an increasing number of biological barriers as the contaminant is transported from roots to shoots to fruits (Blaine et al., 2014; Lesmeister et al., 2021). However, there are other large differences between uptake factors in measurements from different types of plants. Researchers have hypothesized that reasons for these differences may include differences in protein content, differences in root system types and surface areas, the amount of water transpired, the presence of precursors in soil, or the soil conditions where the plants were grown (Ghisi et al., 2019; Lesmeister et al., 2021).

In field studies at locations where biosolids had been applied, there does not appear to be a significant or consistent correlation between PFOA and PFOS uptake factors and soil concentration, pH, organic matter content or cation exchange capacity (Simones et al., 2023). The use of contaminated irrigation water increases uptake of PFOA and PFOS in plants (Gredelj et al., 2020; Blaine et al., 2014). PFAS precursors commonly found in biosolids, such as diPAPs, result in increased perfluorocarboxylic acid concentrations in plants, including PFOA concentrations (Lee et al., 2013; Bizkarguenaga et al., 2016b). More information on the literature search strategy and selected studies used to parameterize PFOA and PFOS uptake factors for the fate and transport models used in this risk assessment is found in Section 2.9.3.

2.6 Effects on Humans and Aquatic and Terrestrial Biota

2.6.1 Human Health Effects

Biosolids risk assessment can consider human health effects that occur after oral, inhalation, or dermal exposures. Due to potential differences in toxicity across oral, inhalation, and dermal exposure pathways, the EPA develops different toxicity values for each pathway.

2.6.1.1 Oral

Based on animal toxicology and human epidemiology studies, oral exposure to either PFOA or PFOS is associated with numerous adverse health effects, including several types of cancer. Through conducting a systematic review of the literature, PFOA and PFOS are relatively high potency PFAS, with very low noncancer reference doses. A detailed description of the health effects observed at various levels of PFOA or PFOS exposure can be found in EPA's recently published Final Human Health Toxicity Assessments (US EPA, 2024b;c).

For PFOA, EPA's toxicity assessment concludes that overall, the available *evidence indicates* that PFOA exposure is likely to cause hepatic, immunological, cardiovascular, and developmental effects in humans, given sufficient exposure conditions (*e.g.*, at serum levels in humans as low as 1.1 to 5.2 ng/mL and at doses in animals as low as 0.3 to 1.0 mg/kg/day)(US EPA, 2024b). These judgments are based on data from epidemiological studies of infants, children, adolescents, pregnant individuals, and non-pregnant adults, as well as short-term (28-day), subchronic (90-day), developmental (gestational), and chronic (2-year) oral-exposure studies in rodents. For hepatic effects, the primary support is evidence of increased alanine transaminase (ALT) levels in humans and coherent evidence of hepatotoxicity in animals, including increased liver weights and hepatocellular hypertrophy accompanied by necrosis, inflammation, or increased liver enzyme levels marking liver injury. For immunological effects, the primary support is evidence of decreased antibody response to vaccination against tetanus, diphtheria and rubella in children, and evidence of immunotoxicity in rodents, including decreased Immunoglobulin M response to sheep red blood cells, reduced spleen and thymus weights, changes in immune cell populations, and decreased splenic and thymic cellularity. For cardiovascular effects, the primary support is evidence of increased serum lipids levels in human and alterations to lipid homeostasis in animals. For developmental effects, the primary evidence is decreased birth weight in human infants and decreased offspring survival, decreased fetal and pup weight, delayed time to eye opening, and related pre- and post-natal effects in animals.

The PFOA toxicity assessment also concludes, consistent with EPA's Guidelines for Carcinogen Risk Assessment (US EPA, 2005a), that the weight of the evidence across epidemiological, animal toxicological, and mechanistic studies indicate PFOA is *Likely to Be Carcinogenic to Humans* via the oral route of exposure. Epidemiological studies provided evidence of kidney and testicular cancer in humans and some evidence of breast cancer in susceptible subpopulations. Chronic oral animal toxicological studies in Sprague-Dawley rats reported Leydig cell tumors, pancreatic acinar cell tumors, and hepatocellular tumors. PFOA exposure is associated with multiple key characteristics of carcinogenicity (Smith, 2016). Available mechanistic data suggest that multiple human relevant modes of action could be involved in the renal, testicular, pancreatic, and hepatic tumorigenesis associated with PFOA exposure in humans and animal models.

For PFOS, EPA's Final Toxicity Assessment concludes the available evidence indicates that PFOS exposure is likely to cause hepatic, immunological, cardiovascular, and developmental effects in humans, given sufficient exposure conditions (*e.g.*, at serum levels in humans as low as 0.57 to 5.0 ng/mL and at doses in animals as low as 0.0017 to 0.4 mg/kg/day). These judgments are based on data from epidemiological studies of infants, children, adolescents, pregnant individuals, and non-pregnant adults, as well as short-term (28-day), subchronic (90-day), developmental (gestational), and chronic (2-year) oral-exposure studies in rodents. For hepatic effects, the primary support is evidence of increased ALT levels in humans and coherent evidence of hepatotoxicity in animals, including increased liver weights and hepatocellular hypertrophy accompanied by necrosis, inflammation, or increased liver enzyme levels marking liver injury. For immunological effects, the primary support is decreased antibody response to vaccination against tetanus, diphtheria, and rubella in children, and evidence of immunotoxicity in rodents, including decreased plaque forming cell response to sheep red blood cells, extramedullary hematopoiesis in the spleen, reduced spleen and thymus weights, changes in immune cell populations, and decreased splenic and thymic cellularity. For cardiovascular effects, the primary support is evidence of increased serum lipids levels in humans and alterations to lipid homeostasis in animals. For developmental effects, the primary support is evidence of decreased birth weight in humans and decreased fetal and maternal weight in animals.

The PFOS Toxicity Assessment also concludes that the weight of evidence across epidemiological and animal toxicological studies indicates that PFOS is *Likely to Be Carcinogenic to Humans* via the oral route of exposure. Epidemiological studies provided evidence of bladder, prostate, liver, kidney, and breast cancers in humans, although evidence was limited or mixed for some cancer types. Findings from chronic oral animal toxicological studies supported findings from human studies. Bioassays conducted in rats reported hepatocellular tumors, pancreatic islet cell tumors, and thyroid follicular cell tumors. Some studies observed multi-site tumorigenesis (liver and pancreas) in male and female rats. PFOS exposure is associated with multiple key characteristics of carcinogenicity (Smith et al., 2016). Available mechanistic data suggest that multiple human relevant modes of action could be involved in pancreatic and hepatic tumorigenesis associated with PFOS exposure in animal models.

These assessments include the derivation of chronic reference doses (RfDs) and cancer slope factors (CSFs). Chronic RfDs are defined as an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily oral exposure for a chronic duration (up to a lifetime) to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime. CSFs are defined as an upper bound, approximating a 95% confidence limit, on the increased cancer risk from a lifetime oral exposure to an agent. RfDs and CSFs are calculated to be protective of the most sensitive effects with the strongest supporting evidence (*i.e.*, those occurring in the lower dose range, also called co-critical effects) relevant to the entire lifespan, including sensitive life stages such as development and pregnancy. For PFOA, the noncancer co-critical effects include reduced antibody response to vaccinations in children (diphtheria and tetanus) (Budtz-Jorgensen & Grandjean, 2018); decreased birth weight (Wikstrom et al., 2020); increased serum total cholesterol (Dong et al., 2019) and the cancer critical effect is increased risk of renal cell carcinoma (Shearer et al., 2021). The noncancer co-critical effects associated with oral exposure to PFOS include decreased birth weight (Wikström et al., 2020); increased serum total cholesterol (Dong et al., 2019) and the cancer critical effect is increased incidence of combined hepatocellular adenomas and carcinomas (Thomford, 2002; Butenhoff et al., 2012).

As at least one of the co-critical effects identified for PFOA and PFOS are a developmental endpoint and can potentially result from a short-term exposure during critical periods of development (in this case, exposure during pregnancy and early life). The EPA concludes that the RfDs for PFOA and PFOS are applicable to both short-term (from 1 to 30 days) and chronic (lifetime) exposure scenarios.

Table 3. Toxicity Values for PFOA

Toxicity Value Type	Value	Critical Effect(s), Critical Study/Studies
RfD (based on epidemiological data)	3 x 10^{-8} mg/kg/day	Reduced antibody response to vaccinations in children (diphtheria and tetanus) (Budtz-Jorgensen & Grandjean, 2018); decreased birth weight (Wikstrom et al., 2019); increased serum total cholesterol (Dong et al., 2019)
CSF (based on epidemiological data)	29,300 $(mg/kg/day)^{-1}$	Renal cell carcinoma (RCC) (Shearer et al., 2021)

Table 4. Toxicity Values for PFOS

Toxicity Value Type	Value	Critical Effect(s), Critical Study/Studies
RfD (based on epidemiological data)	1 x 10^{-7} mg/kg/day	Decreased birth weight (Wikstrom et al., 2019); increased serum total cholesterol (Dong et al., 2019)
CSF (based on animal toxicological data)	39.5 $(mg/kg/day)^{-1}$	Combined hepatocellular adenomas and carcinomas in female rats (Thomford, 2002; Butenhoff et al., 2012, 1276144)

2.6.1.2 Inhalation

The EPA has not completed an assessment of health effects caused by inhalation exposure to PFOA and PFOS. Since an inhalation toxicity value is not available from the EPA or another federal agency, any modeled volatilization of PFOA or PFOS would lead to inhalation exposures that could not be assessed for risk. Also, it is not clear that vapor pressure or Henry's law constants are sufficient to model volatilization of PFOA or PFOS from soil. Surfactants adhere to interfaces and parameters like Henry's law constant are meant for chemicals that reside mainly within water. Furthermore, ionized compounds are commonly less likely to volatilize rapidly and PFOA and PFOS will mainly be in their ionized phase in most farm soils due to the chemicals' acid dissociation constant (pKa) values. No measured data was found to benchmark volatility estimates of PFOA or PFOS from farm soil systems, forested soil systems, lagoons, or sewage sludge monofills. For these reasons, inhalation of PFOA and PFOS are not included as pathways for exposure in the biosolids assessments.

2.6.1.3 Dermal

The EPA's Final Toxicity Assessments for PFOA and PFOS include some discussion of the dermal toxicity and dermal absorption for PFOA and PFOS in humans (US EPA, 2024b;c). ATSDR (2021) also includes some discussion of dermal toxicity and dermal absorption in their Toxicological Profile for Perfluoroalkyls. Neither assessment includes the derivation of a hazard value for direct-contact skin effects or provides a conclusive estimate for dermal absorption rates of PFOA or PFOS. Animal studies of dermal absorption for PFOA indicate that absorption rates of PFOA are impacted by the pH of the exposure media, with highly acidic media and mostly protonated PFOA resulting in higher dermal absorption than less acidic exposure media (ATSDR 2021). There is not expected to be significant dermal absorption of PFOA or PFOS associated with swimming or bathing in waters at normal environmental pH (ATSDR 2021). Though there are uncertainties regarding PFOA and PFOS absorption through dermal soil exposure, at this time there is insufficient information to quantify risk from dermal exposures in the biosolids assessment.

2.6.2 Ecological Effects

2.6.2.1 Effects on Aquatic Organisms

The EPA published Final Aquatic Life Ambient Water Quality Criteria (AWQC) for PFOA and PFOS in October, 2024 (US EPA, 2024l;m). These national recommended criteria represent the highest concentrations of PFOA and PFOS in water that are not expected to pose a risk to the majority (*i.e.*, 95%) of freshwater genera from acute and chronic exposures.

The EPA's final aquatic life AWQCs for PFOA finds that aquatic ecotoxicity data are readily available for freshwater fish, aquatic invertebrates, plants, and algae. Section 3 and Section 4 in the *Final Aquatic Life Ambient Water Quality Criteria for Perfluorooctanoic Acid (PFOA)* provide study summaries of individual publicly available aquatic life studies, and Appendix A through Appendix H of that document summarize the current PFOA aquatic life ecotoxicity data (US EPA, 2024l;m). The mechanisms underpinning the toxicity of PFOA to aquatic organisms is an active and on-going area of research. Additional research is still needed from a mechanistic perspective to better understand how the different modes of action elicit specific biological responses. Molecular disturbance at the cellular and organ level resulting in effects on reproduction, growth and development at the individual level are associated with the sex-related endocrine system; thyroid-related endocrine system; and neuronal, lipid, and carbohydrate metabolic systems (see Ankley et al., 2020 and Lee et al., 2020 for the latest reviews on the subject). The underlying mechanisms of PFOA toxicity to aquatic animals, and fish in particular, appear to be related to oxidative stress, apoptosis, thyroid disruption, and development-related gene expression (Lee et al., 2020). The published research suggests that many of these molecular pathways interact with each other and could be linked. For example, for several PFAS including PFOA, oxidative stress appears correlated with effects on egg hatching and larval formation, linking reproductive toxicity, oxidative stress, and developmental toxicity (Lee et al., 2020). The actual mechanism(s) through which PFAS induce oxidative stress require additional study, but increased ß-oxidation of fatty acids and mitochondrial toxicity are proposed triggers (Ankley et al., 2020).

Of particular importance is that PFOA exposure-related disruption of the sex-related endocrine system (*e.g*., androgen and estrogen) at the molecular, tissue, and organ levels appears to have adverse reproductive outcomes in fish and invertebrates, and likely in both freshwater and saltwater and via multiple exposure routes, *i.e*., waterborne and dietary (Lee at al., 2020). The reproductive effects were observed in the F_0, F_1 and F_2 generations of zebrafish, *Danio rerio*, in the multi-generational PFOA exposure reported by Lee et al. (2017). PFOA causes a wide range of adverse effects in aquatic organisms, including reproductive failure, developmental toxicity, androgen, estrogen and thyroid hormone disruption, immune system disruption, and neuronal and developmental damage.

The published Final Aquatic Life Ambient Water Quality Criteria finds that PFOS ecotoxicity studies are readily available for fish, aquatic invertebrates, plants, and algae. Fewer studies are available for aquatic-dependent birds, reptiles, and mammals; these taxa are not represented in Aquatic Life Ambient Water Quality Criteria and studies on these taxa were not reviewed in EPA's most recent criteria. Sections 3 and 4 of the *Final Aquatic Life Ambient Water Quality Criteria for Perfluorooctane Sulfonate (PFOS)* provide study summaries of individual, publicly available aquatic life toxicity studies, and Appendix A through Appendix H of that document summarize current PFOS aquatic life ecotoxicity data (US EPA, 2024l;m). PFOS is one of the most studied PFAS in the ecotoxicity literature, with reported adverse effects on survival, growth, and reproduction. However, additional research is needed to better understand the modes of action of PFOS. Specifically, additional research from a mechanistic perspective is needed to better understand how the different modes of action elicit specific biological responses in fish, aquatic invertebrates, and amphibians. Potential effects of PFOS involving multiple biological pathways are a research challenge for PFOS. Toxicity literature indicate that PFOS causes a wide range of adverse effects in aquatic organisms, including reproductive effects, developmental toxicity, and estrogen, androgen and thyroid hormone disruption (see Sections 3 and 4 and Appendices A.1 through H.1; US EPA, 2024l;m). Following exposure to PFOS, molecular level events can perturb estrogen-, androgen- and thyroid-related endocrine systems, as well as neuronal, lipid, and carbohydrate metabolic systems and lead to cellular- and organ-level disturbances and ultimately result in effects on reproduction, growth, and development at the individual organism level (Ankley et al., 2020; Lee et al., 2020). The mechanisms of PFOS toxicity to fish in particular appear to be related to oxidative stress, apoptosis, thyroid disruption, and alterations of gene expression during development (Lee et al., 2020). Notably, PFOS exposure appeared to be related to the disruption of the sex hormone-related endocrine system at the molecular, tissue, and organ levels, resulting in observed adverse reproductive outcomes in freshwater and saltwater fish and invertebrates alike. Further, these effects have been reported after exposure via multiple exposure routes (*i.e.*, waterborne, dietary, maternal; Lee et al. 2020). And these reproductive effects also appeared to be trans-generational, as observed in a multi-generational zebrafish (*Danio rerio*) study by Wang et al. (2011a).

The EPA established the national recommended criteria for PFOA and PFOS to be protective of most aquatic organisms in the community (*i.e.*, approximately 95 percent of tested aquatic organisms representing the aquatic community). The criteria are protective of aquatic life designated uses for freshwaters. The PFOA and PFOS criteria documents contain acute and chronic criteria for freshwaters (see Table 5). The criteria documents also contain chronic criteria expressed as tissue-based concentrations to protect aquatic life from PFOA and PFOS bioaccumulation. The chronic freshwater and chronic tissue criteria are intended to be independently applicable and no one criterion takes primacy. The criteria reflect the maximum concentrations, with associated frequency and duration specifications, that would support protection of aquatic life from acute and chronic effects associated with PFOA and PFOS in freshwaters.

Table 5. Freshwater Aquatic Life AWQCs for PFOA and PFOS

Criteria Component	Acute Water Column (CMC)[1]	Chronic Water Column (CCC)[2]	Invertebrate Whole-Body	Fish Whole-Body	Fish Muscle
PFOA Magnitude	3.1 mg/L	0.1 mg/L	1.18 mg/kg ww	6.49 mg/kg ww	0.132 mg/kg ww
PFOS Magnitude	0.071 mg/L	0.00025 mg/L	0.028 mg/kg ww	0.201 mg/kg ww	0.087 mg/kg ww
Duration	1-hour average	4-day average	Instantaneous[3]		
Frequency	Not to be exceeded more than once in three years, on average	Not to be exceeded more than once in three years, on average	Not to be exceeded		

[1] Criterion Maximum Concentration.

[2] Criterion Continuous Concentration.

[3] Tissue data provide instantaneous point measurements that reflect integrative accumulation of PFOA or PFOS over time and space in aquatic life population(s) at a given site.

2.6.2.2 Effects on Terrestrial Organisms

Plant and terrestrial vertebrate studies are typically focused on mortality, reproduction, development, or growth effects that would impact a large fraction of the population. Studies on these organisms that are sub-lethal are less commonly available, especially for plants and terrestrial vertebrates. As a result, more sensitive adverse endpoints in wildlife may not be observed, even if they do exist. These factors may lead to hazard values that are higher (indicative of lower toxicity) than studies measuring effects at the individual organism level.

Plants: There are no existing federal assessments that describe the phytotoxicity of PFOA and PFOS, though there are several journal publications on the topic. Tests to find the 50% inhibition concentration (the contaminant concentration that causes 50% of the inhibition effect in organism growth, or IC$_{50}$) of PFOA and PFOS tend to find results ranging from the 10's to 10,000's µM, which are significantly higher than concentrations typically found in the environment (Li et al., 2022). The phytotoxicity of direct soil exposure to *Brassica chinensis* root growth after a 7-day exposure to PFOA and PFOS in six different soils was evaluated (Zhao et al., 2011). The 50% effect concentration for root elongation (EC$_{50}$) values ranged from 95 mg/kg to >200 mg/kg for PFOS and from 107 mg/kg to 246 mg/kg for PFOA. In a study by Brignole et al. (2003), the effects of PFOS on the seedling emergence and growth of seven species of plants was evaluated after a 21-day exposure. Lettuce (*Lactuca sativa*) was the most sensitive species tested with a 25% effect concentration (EC$_{25}$) of 6.79 mg/kg, based on seedling height. The EC$_{25}$s for the other six plant species were: 7.51 mg/kg (ryegrass), 11.7 mg/kg (tomato), 12.9 mg/kg (onion), 53.3 mg/kg (alfalfa), 81.6 mg/kg (flax), and 160 mg/kg (soybean), all based on shoot weight.

Invertebrates: Toxicity values for direct soil exposure to earthworms (*Eisenia fetida*) have been determined for PFOA and PFOS. The 14-day 50% lethal concentration (LC$_{50}$) values for earthworms exposed to a loamy sandy soil spiked with varying concentrations of PFOA and PFOS were determined to be 811 mg/kg and 541 mg/kg, respectively (Yuan et al., 2017). PFOS toxicity values are also available for two additional invertebrate species, *Folsomia candida* (springtail) and *Oppia nitens* (oribatid mite) (Princz et al., 2018). Springtails and oribatid mites were exposed to PFOS in two types of soil: a coarse-textured sandy loam and fine-textured clay loam. The 25% inhibition concentration (IC$_{25}$) values, based on juvenile reproduction, for oribatid mites were 13 mg/kg and 33 mg/kg in the fine and coarse soil, respectively. For springtails, the IC$_{25}$s were 74 mg/kg and 185 mg/kg for the fine and coarse soil, respectively.

Birds: To date, a limited number of laboratory studies have been conducted on a small number of bird species to determine the toxicity of PFAS. The 50% lethal dose (LD_{50}) values for juvenile mallard ducks (*Anas platyrhynchos*) and northern bobwhite quail (*Colinus virginianus*) fed for five days with PFOS in their diet were determined to be 150 mg PFOS/kg bw/day and 61 mg PFOS/kg bw/day, respectively (Newsted et al., 2006). For Japanese Quail (*Coturnix japonica*) fed PFOS and PFOA in their diet for five days, the LD_{50}s were 38 mg/kg bw/day and 68 mg/kg bw/day for PFOS and PFOA, respectively (Bursian et al., 2021). A chronic laboratory study examined the adult health, body and liver weights, feed consumption, gross morphology and histology of body organs, and reproduction in adult mallard ducks and bobwhite quail exposed to PFOS in their diet for 21 weeks (Newsted et al., 2007). For bobwhite quail and mallard ducks exposed to 50 and 100 mg PFOS/kg feed, lethality was observed within five weeks from the onset of exposure, whereas no effects on survival were observed in the 10 mg PFOS/kg feed treatment. In the 10 mg PFOS/kg treatment groups, no significant effects were noted in mallard ducks. However, the lowest observable adverse effect level (LOAEL) was determined to be 10 mg/kg PFOS in feed based on decreased survivorship of 14-day-old bobwhite quail offspring.

In 2018, ECCC published Federal Environmental Quality Guidelines (FEQGs) for PFOS (ECCC 2018). These FEQGs are benchmarks for the quality of the ambient environment that are based solely on the toxicological effects or hazards posed by substances. ECCC identified the quail survivorship study as a critical study for effects in birds and calculated a bird egg FEQG of 1.9 ug/g ww. The assessment also notes a field study compared reproductive success in tree swallows from a contaminated urban lake versus a reference lake (Custer et al., 2012). The authors concluded that PFOS concentrations above 0.15 µg/g egg were detrimental to hatching success; however, the FEQG authors state that this study could not be considered in FEQG development because of variability in hatch success between the two field seasons, variations in egg PFOS concentrations within clutches, and concurrent exposure to other PFAS. More information is needed on adverse impacts of PFOS to birds.

Livestock and game: A recent review paper (Death et al. 2021) summarizes the literature on toxic effects of PFOA and PFOS in livestock and wild game. Studies measuring the uptake, elimination, and distribution of PFOA and PFOS in various livestock have not reported adverse effects in the test animals (Wilson et al. 2020, Vestegren et al. 2013, Numata et al., 2014). Death et al. (2021) similarly finds that while there are multiple studies identifying PFOA and PFOS occurrence in wild game (ducks, deer, wild boar), these studies have not identified adverse effects in the game associated with PFOA and PFOS exposure.

2.6.3 Scoping: Sensitive Receptors and Endpoints

Overall, adverse effects observed in plants, invertebrates, fish, and birds occur at concentrations that are higher than levels that would be associated with adverse effects in humans; there have been no studies reporting adverse effects occurring in livestock. Therefore, this draft risk assessment has been scoped to focus on *human health* risks. Below is a brief comparison of the inherent toxicity of PFOA and PFOS to humans versus other organisms.

Soil. As described above in Section 2.6.2.2, adverse effects observed in plant studies occur at soil concentrations ranging from the 10’s to 100’s of mg/kg PFOA and PFOS. Similar ranges of effect levels in soils are found for terrestrial invertebrates, where the effect levels for terrestrial invertebrates are in the 10’s to 100’s of mg/kg PFOA or PFOS. Concentrations of PFOA or PFOS in soil that are protective of human health through incidental ingestion are expected to be significantly lower than the effect levels observed for plants and terrestrial invertebrates. For example, risk-based thresholds for PFOA and PFOS in soils protecting against non-cancer effects in children are on the order of 0.001-0.010 mg/kg PFOA or PFOS in soil (US EPA, 2024i).

Surface water. The thresholds established in EPA's Final Aquatic Life AWQCs protective of fish are higher (less stringent) than would be expected to be protective of human consumers of home-caught fish. The EPA is developing national recommended human health criteria for PFOA and PFOS, based on the agency's final toxicity assessments (US EPA 2024b,c) which would take into account exposures via drinking water, fish consumption, and other sources (*e.g*., other dietary sources, consumer products, etc.). The most stringent EPA national recommended (chronic) aquatic life criteria for PFOA and PFOS are 1.0 x 10^{-1} mg/L (100 ug/L) for PFOA and 2.5 x 10^{-4} mg/L (0.25 ug/L) for PFOS; State surface water standards to protect human health due to fish consumption have established values that are lower than the EPA's Aquatic Life AWQC for PFOA and PFOS. For example, the state of Minnesota has established surface water criteria protective of non-cancer effects in fish consumers that are 8.8 x 10^{-5} mg/L (88 ng/L) for PFOA (MPCA, 2023b) and 5 x 10^{-8} mg/L (0.05 ng/L) for PFOS (MPCA, 2020). Fish tissue thresholds protective against cancer effects in human fish consumers would be lower (more protective) than those developed in Minnesota. Further, some surface waters are used as a source of drinking water. Risk-based thresholds for PFOA and PFOS in drinking water are also lower than the aquatic life criteria (less than 5 x 10^{-6} mg/L, *5 ng/L,* for PFOA and PFOS, US EPA 2024b;c).

These trends of human health-based thresholds being more stringent than ecologically protective thresholds are evident due to the extremely potent nature of PFOA and PFOS toxicity in humans. More study of PFOS and PFOA effects in wildlife could result in a narrowing of the gap between levels protective of ecological endpoints and levels protective of human health. However, based on currently available data, the EPA is focusing on human health endpoints for the biosolids draft risk assessment, with the understanding that establishing practices protective of human health will also offer protection to aquatic life, terrestrial wildlife, and livestock health.

If future studies indicate ecological toxicity of PFOA or PFOS at lower doses/concentrations (*e.g*., for terrestrial organisms), the EPA may conduct further ecological risk assessment, as warranted. This human health focused draft risk assessment for PFOA and PFOS does not preclude any future biosolids-related unacceptable risk finding for aquatic life, terrestrial wildlife, or livestock.

2.7 Exposure Pathways for Humans and Aquatic and Terrestrial Biota

As described in Section 1.3, sewage sludge can be disposed of via solid waste landfill, surface disposal at a dedicated sewage sludge disposal site, or incineration, or it can be land applied as a soil amendment to a variety of sites (agricultural fields, public access areas, road construction, landfill cover material, soil material in remediation efforts, and more). These disposal and land application options all result in potential pathways for PFOA and PFOS exposure to humans and wildlife, such as drinking water consumption, dietary intake, soil ingestion, and inhalation of particulate-bound contaminants.

Currently, there is insufficient information available to model occupational exposures for workers that repeatedly apply biosolids at different farms throughout the year or to determine whether the farm family or farm worker exposures will exceed the exposures of these professional biosolids applicators. If this type of worker is repeatedly spray applying biosolids on farm fields, that could lead to airborne exposures over many days of the year and this type of exposure is not represented within the modeled pathways for the farm family. The EPA also does not currently have survey or other data to estimate the behavior patterns of these types of workers with missing information including amount of biosolids mass aerosolized during application, time spent per day applying biosolids, and number of days worked per year. As mentioned above, there is not currently a reference concentration (RfC) or inhalation unit risk (IUR) available for PFOA or PFOS, so the risks this type of worker may face cannot be assessed due to an absence of exposure and toxicity values. For these reasons, this draft risk assessment does not include receptors of professional biosolids land appliers in the conceptual models.

2.7.1 Considerations of Aggregate Exposures

Aggregate exposure and risk assessment involve the analysis of exposure to a single chemical by multiple pathways and routes of exposure. This draft risk assessment does not aggregate exposure and risk, and instead presents estimated exposure and risk for each individual exposure pathway (*i.e.*, consuming fish, drinking water, incidentally ingesting soil). This approach does not account for exposure from multiple modeled pathways simultaneously, pathways that were not modeled due to data gaps (including inhalation and dermal exposure pathways) or exposure pathways not related to sewage sludge use and disposal (such as exposure from use of personal care products, cleaning supplies, household dust, etc.). This decision to assess each pathway individually allows modeling results to be interpreted as risk contributed from sewage sludge for each pathway across a variety of sewage sludge use and disposal scenarios.

Assessing individual pathways also allows risk assessors to consider a variety of potential receptors who may have exposure from some, but not all of the potentially relevant exposure pathways. However, in each given scenario, a receptor may be exposed from multiple pathways at the same time and from pathways not modeled in this risk assessment. For example, farmers who consume animal products produced on the farm likely also consume drinking water sourced locally as many rural areas of the country rely on groundwater as a source of drinking water.

2.7.2 Considerations of Cumulative Exposures

Cumulative exposure and risk assessment involve analysis of exposures from multiple stressors that occur simultaneously. A receptor may be exposed to both PFOA and PFOS at the same time. PFOA and PFOS have been shown to be dose additive (US EPA, 2024e) and are nearly always found in mixtures in sewage sludge. It follows that the environmental media impacted by use or disposal of sewage sludge also contains mixtures of PFOA and PFOS. The presence of mixtures and multiple pathways for exposure will result in higher risks of adverse health effects at a population scale than are reflected in the pathway-specific results. This draft risk assessment presents exposures and risks (hazard quotients and cancer risk levels) associated with single chemicals (PFOA or PFOS) to provide information about which compound is contributing most significantly to exposure and risk in each pathway. Though this draft risk assessment is scoped narrowly to PFOA and PFOS, other PFAS are also known to be present in biosolids (see Section 2.4), and the EPA may consider additional PFAS for risk assessment in the future.

2.8 Conceptual Models

There are a multitude of potential unique strategies and hydrogeochemical settings for biosolids disposal and reuse across the U.S. It is not feasible to model or assess each of these environmental release scenarios individually. Instead, the EPA has strategically selected a discrete number of common reuse or disposal scenarios to model in detail and will use the findings from these detailed modeling exercises to qualitatively describe other relevant scenarios. These detailed modeling scenarios were selected because 1) they are commonly used for sewage sludge, biosolids, or septage in the U.S., 2) they are likely to result in higher exposures for humans, or 3) they include numerous pathways that are applicable to other reuse or disposal scenarios. In some cases, such as biosolids incineration and other uses of biosolids in land application (silviculture, golf courses, etc.), there are data limitations that restrict our ability to quantitatively assess exposure outcomes.

Four detailed modeling scenarios are described in this document: reuse on a farm growing fruits and vegetables (crop farm scenario), reuse on a farm raising livestock (pasture farm scenario), disposal in a surface disposal site (surface disposal scenario), and reuse to restore over-grazed pastureland (land reclamation scenario). These detailed models are used quantitatively to estimate exposure and describe potential risks to human receptors in each scenario. These models are also used to qualitatively estimate relative exposures and risks associated with other types of land application like use in silviculture or application to golf courses, other types of land reclamation like mine reclamation or road construction, and incineration. For each of the modeling scenarios, the EPA conducts modeling runs parameterized for hypothetical regions in a wet climate, a dry climate, and a moderate climate. These region-specific meteorological conditions, soil conditions, and hydrologic conditions are described in Section 2.9.3. These models are not intended to characterize conditions at any specific site.

The following sections illustrate the conceptual models for PFAS application, transport, uptake, and exposure in each disposal and reuse scenario; additional information on the computational models used and the parameterization of those models can be found in Section 2.9 and Appendices B and C. As described in Section 2.6.1, dermal exposures are not expected to meaningfully contribute to overall exposure, and dermal exposure pathways are not included in the conceptual models for this risk assessment. Inhalation is not expected to be a significant source of exposure for these scenarios and there are no inhalation toxicity values (RfCs or IURs) available for PFOA and PFOS; for these reasons, inhalation pathways are also not included in the conceptual models. Finally, data available to date indicate that PFOA and PFOS are significantly more toxic to humans than wildlife or livestock, such that actions taken to protect human health will also protect wildlife and livestock health. The following conceptual models therefore only include exposure pathways relevant to humans.

2.8.1 Farms

Two types of farming scenarios are included in this assessment: a farm growing fruits and vegetables (the crop farm) and a farm raising animals (the pasture farm).

2.8.1.1 Crop Farm Scenario

The crop farming scenario is designed to capture relevant human exposure pathways for PFOA and PFOS following biosolids land application to fields used to grow human food. **Figure 2** provides a schematic visualization of the crop farming scenario. **Figure 3** presents the conceptual model for the crop farm, showing the different pathways evaluated.

Previous biosolids assessments (US EPA 1992, 1995a, 2003a) have assessed this scenario, and the original exposure pathway numbers from the 1993 assessment (*i.e.*, the one conducted to support the 1993 regulation, US EPA 1992) are included in Figure 3 for reference. While some states have regulated the application of biosolids to fields used to grow human food, this practice is not regulated in other states. Furthermore, because of the extreme persistence of PFOA and PFOS in soils, a property with previous biosolids land application that has been repurposed as a farm for human food could still have multiple relevant human exposure pathways. Class A biosolids currently have no restrictions on crop type or harvesting delay restrictions for agricultural applications. Finally, Class A exceptional quality (Class A_{EQ}) biosolids, which can be used by home or hobby gardeners, have no restrictions on their application rates or use to grow food for human consumption, though they do have some restrictions on maximum concentrations of some metals.

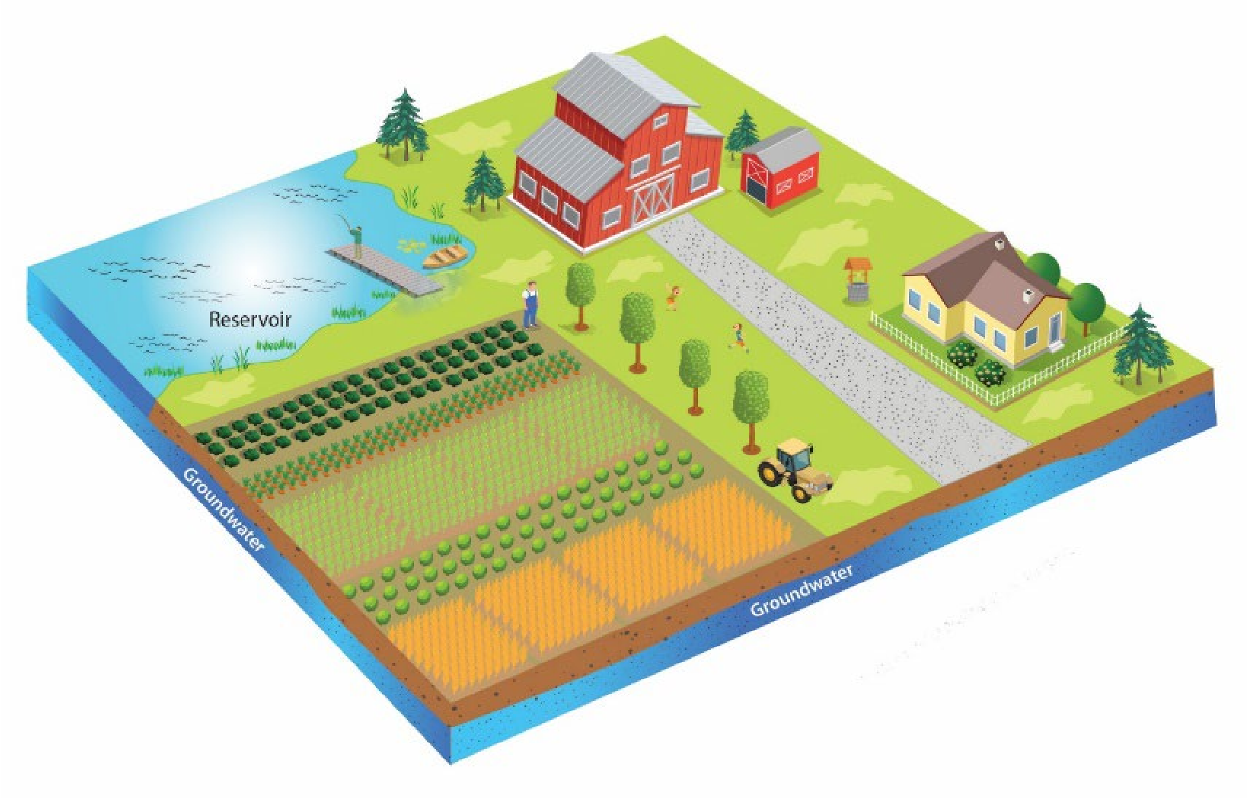
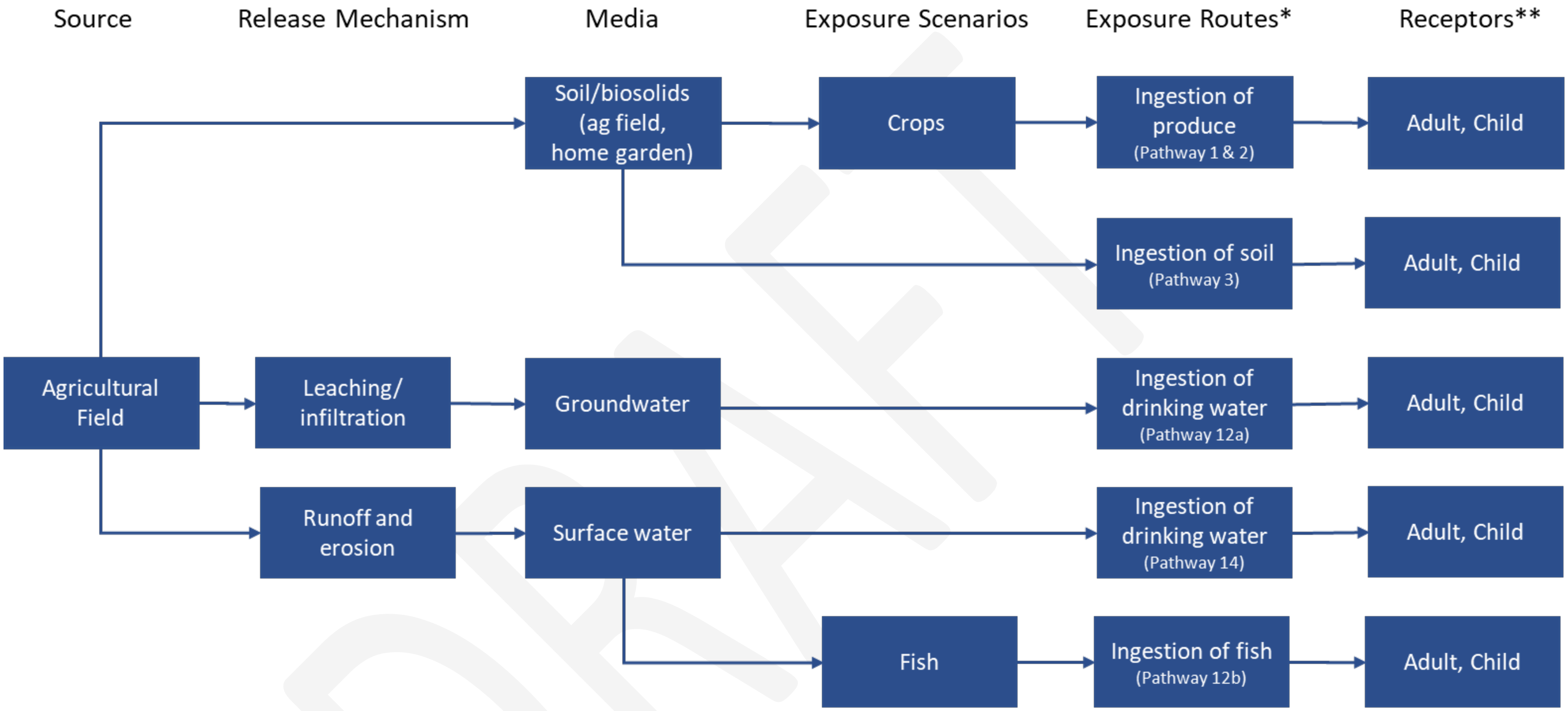


Figure 2. Conceptual visual depiction of crop farming scenario.



* Pathway numbers refer to those in the original 1993 biosolids risk assessment. No pathway number indicates a pathway that has been added since that original assessment.

** Receptor populations may include farm families, home gardeners, CSA participants, or nearby residents.

Figure 3. Crop farm conceptual model

The crop farm model can be applied to several scenarios of biosolids use. First is the model application of biosolids to commercial crop farm or hobby/subsistence farm, where a family lives adjacent to the land used for crop cultivation. In this scenario, adults and children on the farm could have exposure through consuming crops grown on the field, drinking water, and incidental soil ingestion. The crop farm model also includes pathways that could be relevant to neighbors, those supporting the farm through CSA arrangements, or those purchasing food at the family's farm stand. Finally, by evaluating exposure with the non-limited application rates allowed for Class A_{EQ} biosolids, this scenario captures potential impacts to the home gardener from applying biosolids at their personal or community gardens.

The crop farm scenario is important to model quantitatively because it includes receptors that are likely to have higher exposure rates than receptors in other scenarios. For example, a self-sufficient farm family that spends most of the year immediately adjacent to the farm is assumed to be exposed to multiple transport pathways (drinking water, soil, fish, other food) and engage in behaviors that lead to them having higher exposures than the general population (*i.e.*, consuming a high portion of their total produce intake from a single potentially contaminated farm). Community members that purchase large amounts of produce from the farm via CSA or frequent farm stand purchases will also have more of their dietary intake from a single, potentially contaminated source than the general population. A risk assessment of these pathways is therefore also protective of produce consumers in the general U.S. population.

The potential impacts from application of contaminated biosolids to a particular farm site (*e.g.*, field) can have broader implications to the farm's neighbors and the larger community. The use of the farm family as a surrogate to represent other populations means that an assessment of the potentially impacted populations from the land application of biosolids should not be limited to self-sufficient farmers. For example, a farm's neighbors or an entire community might rely on the same drinking water source as the farm family.

After land application of contaminated biosolids, there are multiple potential human exposure pathways. Once biosolids have been applied, PFOA and PFOS will either stay in the soil column of the farm field or garden, move with windblown soil particles, infiltrate through the soil column into groundwater, or mobilize in the particulate or sorbed phase through runoff and erosion into surface water. PFOA and PFOS in the soil on the farm field can be taken up into the edible or non-edible portion of crops. PFOA and PFOS that infiltrate into groundwater will infiltrate to the uppermost aquifer and then flow downgradient with groundwater, where they could end up in well water used for human drinking water. The chemicals transported to a nearby lake or reservoir could be in drinking water or be taken up into edible fish tissues. The PFOA and PFOS in soils is available for child or adult incidental soil ingestion.

This draft risk assessment will focus on potential exposures that result from drinking water ingestion, dietary ingestion, and incidental soil ingestion. Exposure from drinking water ingestion could result from contamination of groundwater following leaching of PFOA and PFOS through soil and from contamination of surface water following erosion and runoff. Exposure from dietary ingestion could include consumption of fish and produce (fruits and vegetables). Soil ingestion exposures are based on incidental soil ingestion values for children from soil on the farm field or gardening area.

The exposure model estimates the most significant transport pathways for chemicals in biosolids, but some less significant pathways are not included. For example, in some scenarios, farmers may use groundwater or surface water that is contaminated by PFOA and/or PFOS as irrigation water, which could result in additional crop uptake of these chemicals and thus potential human exposure. Additionally, the model assumes that the farm field has no PFOA or PFOS present in soils (*e.g.*, via atmospheric deposition) prior to the application of biosolids.

2.8.1.2 Pasture Farm Scenario

The pasture farm scenario is similar to the crop farm scenario but models a farm that only raises animals (cows and chickens) and crops used for livestock feed, rather than fruit and vegetable crops for human consumption. **Figure 4** provides an illustrative visualization of the pasture farming scenario. **Figure 5** presents the conceptual model for the pasture farm, showing the different pathways evaluated.

This scenario has also been considered in prior risk assessments (see pathway numbers in Figure 5). While some states restrict the use of some biosolids on fields used to grow food for human consumption, there are very few states or other jurisdictions that restrict the use of biosolids for fields used to grow feed for animals or fields used for animal grazing (ECOS, 2023). Furthermore, because of the extreme persistence of PFOA and PFOS in soils, a property with previous biosolids land application that has been repurposed as a pasture for animal grazing or field for growing feed would still have multiple potential human exposure pathways available.

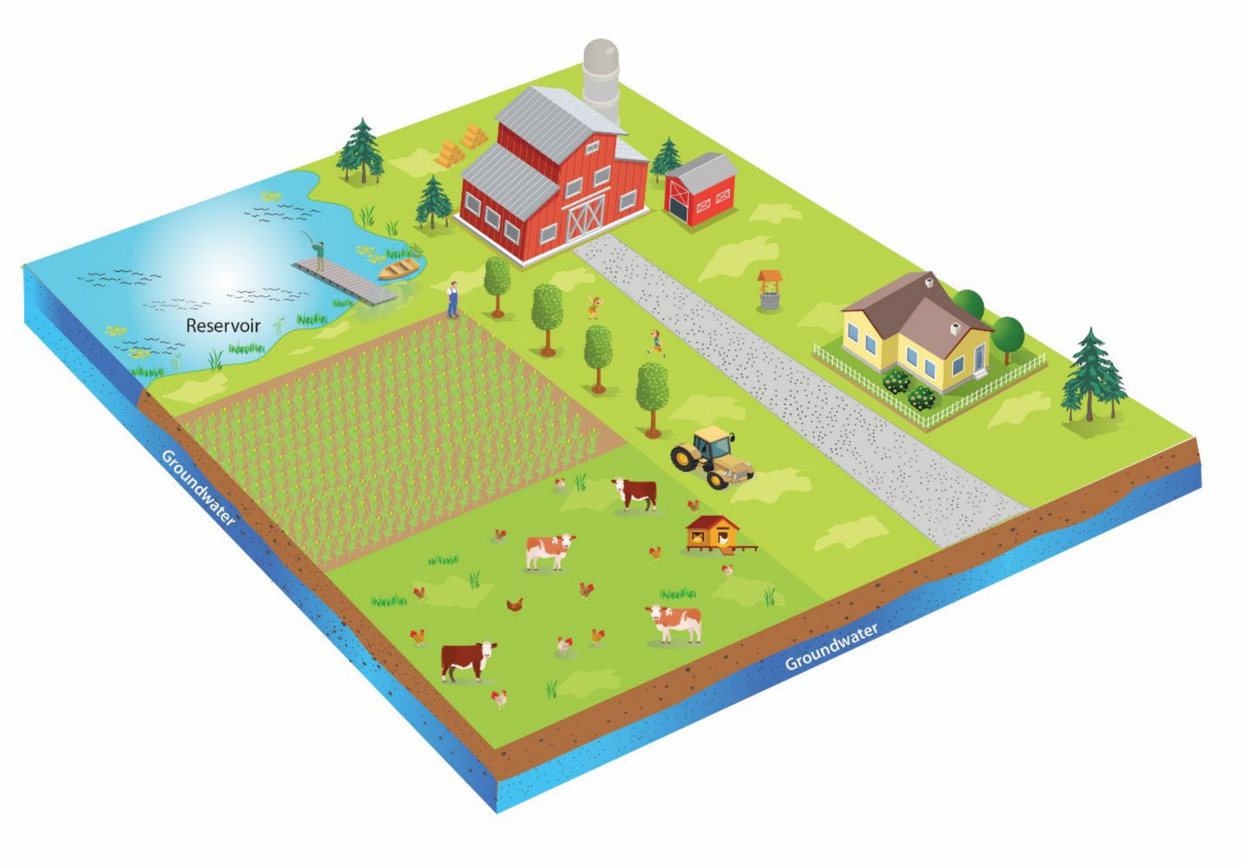
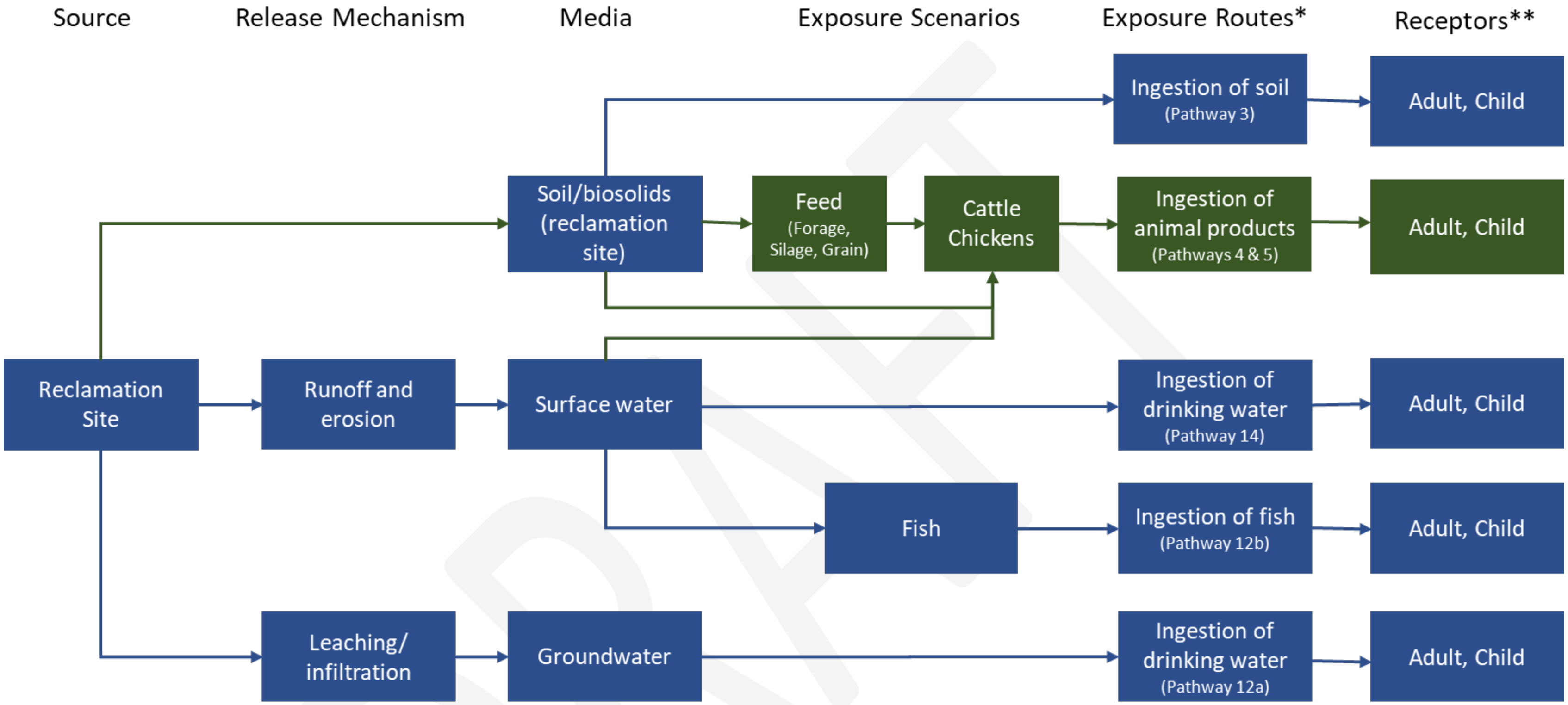


Figure 4. Conceptual visualization of pasture farm scenario.



* Pathway numbers refer to those in the original 1993 biosolids risk assessment. No pathway number indicates a pathway that has been added since that original assessment. Some pathways have been modified or combined; here, pathways 4 (cattle eating plants) and 5 (cattle eating soil) have been combined, and consumption of chicken and eggs has been added.

** Receptor populations may include farm families, home gardeners, CSA participants, or nearby residents.

Green boxes and lines represent farm pathways that may not be present at all reclamation sites.

Figure 5. Pasture farm conceptual model.

The pasture farm model can be applied to several scenarios of biosolids use, similar to the crop farm model. First is the application of biosolids to a commercial or hobby/subsistence farm, where a family lives adjacent to the land used for grazing cows, raising chickens, or growing feed for these animals. In this scenario, adults and children on the farm could have exposure through meat, dairy, or egg products they produce, and incidental soil ingestion in the yard or land near their home. Similar to the crop farm, the pasture farm also includes pathways that could be relevant to neighbors and the larger community, *e.g.*, those sharing a drinking water supply, supporting the farm through CSAs or those frequently purchasing meat, milk, or dairy at a local market or farm stand.

The pasture farm scenario is also important to model quantitatively because it includes receptors that are likely to have higher exposure rates. For example, a self-sufficient farm family that spends most of the year living on the farm may be exposed to multiple transport pathways (drinking water, soil, fish, food) and may engage in behaviors that lead to them having higher exposures than the general population (*i.e.*, consuming a high portion of their total meat, milk, and egg intake from a single source). Farm neighbors or an entire community could use the same drinking water source as the farm family. Community members that purchase large amounts of food from the farm via CSAs and frequent market or farm stand purchases will also have more of their dietary intake from a single, potentially contaminated source, potentially resulting in higher exposures than the general population. A risk assessment of these pathways is therefore also protective of meat, milk, and egg consumers in the general U.S. population.

After land application of biosolids, there are multiple pathways that could cause exposures to humans in the pasture farm model. Unlike the crop farm scenario, in the pasture farm scenario, it is not assumed that biosolids are tilled into the soil. For this reason, once biosolids have been applied, more PFOA and PFOS will be available to move with windblown soil particles or mobilize in the particulate or sorbed phase through runoff and erosion into surface water. The PFOA and PFOS that remain in the soil on the farm field could be taken up into the grass or hay used for animal feed or grazing. In the pasture farm scenario, groundwater and surface water can be used by humans and livestock as a drinking water source. The soil on the field can be consumed by animals foraging or grazing. Other potential pathways relevant to the crop farm scenario (human ingestion of fish and soils) are also relevant to the pasture farm scenario.

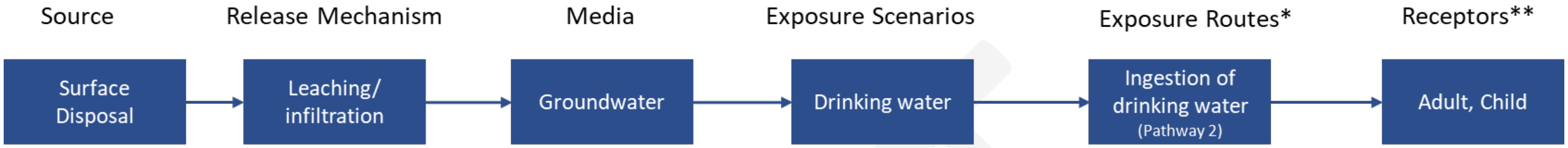
2.8.2 Land Reclamation

One known use of biosolids in the U.S. is for the purpose of increasing the organic matter content in fields that have been over-grazed and have degraded soil quality. Biosolids have been used in these settings as a beneficial soil amendment and may be applied at rates higher than those allowed under traditional agricultural settings. For the purposes of this draft risk assessment, the EPA has modeled the land reclamation scenario using the same conceptual model used for the pasture farm (Figure 5). Though the pathways relevant to the reclamation scenario are the same as those relevant to the pasture farm scenario, the rate of biosolids land application in the reclamation scenario is modeled as higher than the land application rate used for the pasture farm. Additionally, in the reclamation scenario there is only one application of biosolids, instead of ongoing annual applications modeled in the pasture scenario. However, other than the differences in biosolids land application rate and timing, the potential human exposure pathways in this scenario are the same as those in the pasture farm scenario. If a site is being remediated in this fashion to improve soil quality, but is not then used as a pasture, the same conceptual model applies, except that the pathways related to livestock are not relevant.

Land reclamation can take many forms, and no two land reclamation projects are exactly alike. Biosolids have been used in a wide variety of land reclamation settings such as remediating closed mines, remediating soils at clean-up sites with industrial pollution, or amending soils disturbed by new construction. These different types of sites could have a variety of hydrologic, geologic, and geochemical conditions than influence the fate and transport of PFAS. These sites could also have a number of different potential exposure pathways for human exposure. Previous risk assessments have focused on mine reclamation and over-grazed farmland because these activities were judged to be most sensitive reclamation scenarios for the chemicals being modeled (US EPA, 1992; US EPA, 2003a). The EPA is selecting a reclamation scenario of remediating over-grazed farmland for PFOA and PFOS because the accumulation of these chemicals into livestock is likely to represent a higher human health risk scenario for a farmland reclamation relative to other reclamation activities. This scenario also includes some potential pathways (such as the soil to groundwater to drinking water pathway) that are widely applicable across many potential land reclamation settings.

2.8.3 Surface Disposal

Surface disposal is the placement of sewage sludge in an active sewage sludge unit for final disposal, not for treatment, storage, or to condition the soil or fertilize crops grown in the soil. The surface disposal scenario is designed to capture potential human exposure pathways for PFOA and PFOS that are available after sewage sludge is placed in a surface disposal site. **Figure 6** presents the conceptual model for the surface disposal scenario, showing the pathway evaluated.



* Pathway numbers refer to those in the original 1993 biosolids risk assessment.
** Receptor population is nearby residents.

Figure 6. Conceptual model for disposal in a surface disposal site.

The characteristics of surface disposal sites are varied. In some cases, sewage sludge is dewatered and disposed of in a sewage sludge-only landfill (known as a monofill) which can be lined or unlined. In other cases, the sewage sludge is not dewatered before disposal. These surface disposal sites can also be lined or unlined. This draft risk assessment models the potential impacts of PFOA and PFOS migration at lined and unlined surface disposal sites. Surface disposal of sewage sludge that has not been dewatered represents the scenario with the greatest potential for environmental releases via leaching and infiltration, so this specific scenario is modeled in the risk assessment.

Sewage sludge may also be sent to a lagoon. The EPA considers lagoons to be waste stabilization ponds or basins designed and built to reduce organic content, suspended solids, and pathogens in wastewater and sewage sludge. They can be lined or unlined. From a groundwater infiltration perspective, these lagoons are not dissimilar from sewage sludge surface disposal sites accepting materials that have not been dewatered. Though lagoons are a treatment technology, not a disposal method, the modeling exercises in this risk assessment can also be used to qualitatively understand potential infiltration risks at some lagoons.

MSW landfills also receive sewage sludge for disposal along with many other waste streams, but those facilities are outside the scope of CWA section 405 and will not be assessed here as they fall under the regulations of Resource Conservation and Recovery Act (RCRA) Subtitle D. Similarly, the use of sewage sludge as daily cover on MSW landfills is also relevant to RCRA. The EPA has provided information on MSW landfills for the disposal of sewage sludge and other PFAS-containing wastes in its Interim Guidance on Destroying and Disposing of Certain PFAS and PFAS-Containing Materials That Are Not Consumer Products (US EPA, 2024g).

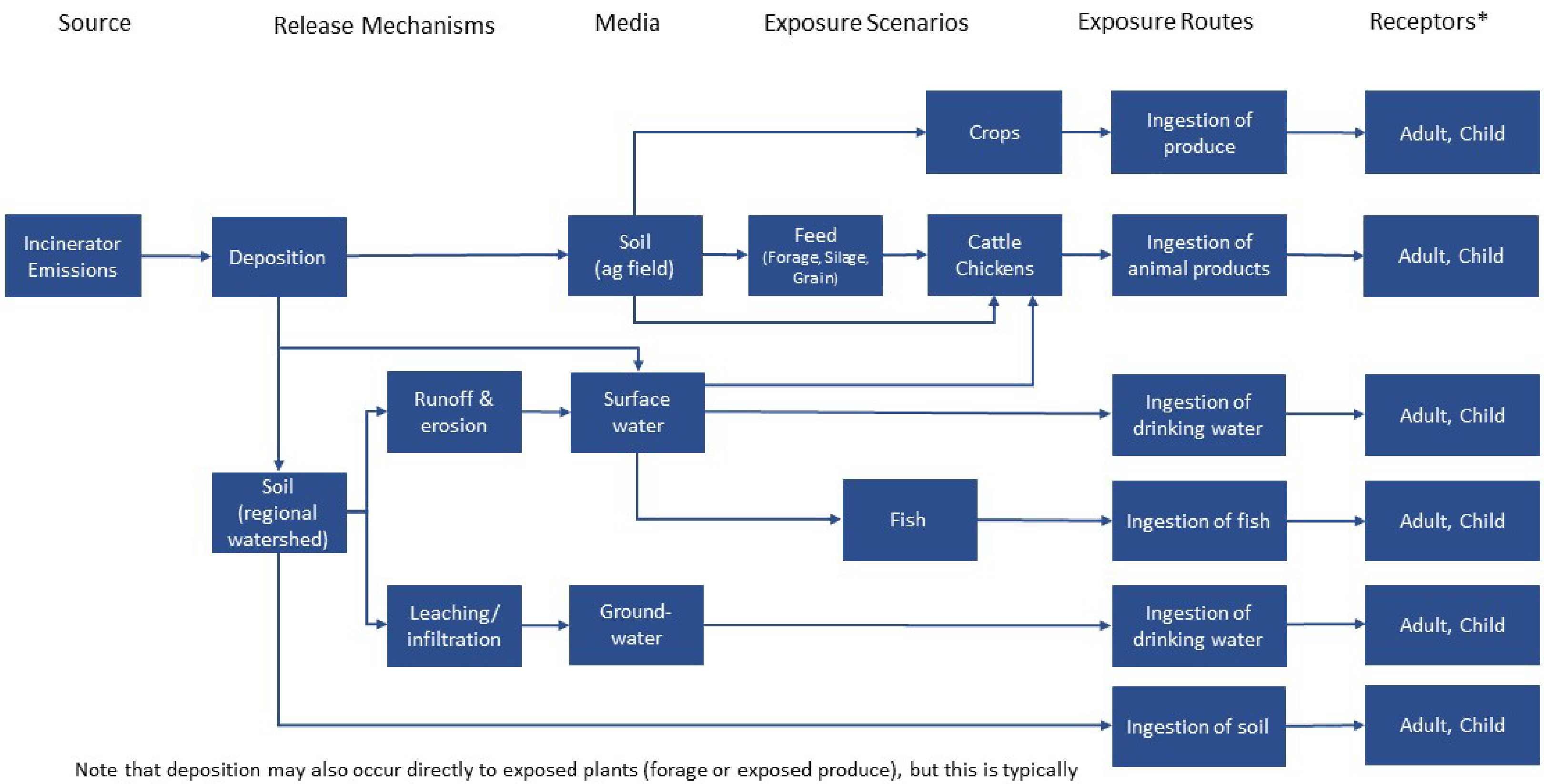
2.8.4 Incineration

The incineration model (**Figure 7**) illustrates PFOA and PFOS exposure pathways that are possible after sewage sludge is incinerated in an SSI. Contaminant levels for sewage sludge entering an SSI are regulated by the CWA via part 503, and emissions from SSIs are regulated under the CAA (US EPA, 2023b), but the CAA regulations do not currently include any requirements related to PFOA or PFOS.

SSIs are devices used for the combustion of dewatered sewage sludge. In the U.S., the two main types of SSIs include multi-hearth furnaces and fluidized bed combustors (US EPA, 2016). In a multi-hearth furnace, the sludge is typically dried at temperatures ranging from 425°C to 760°C (800°F to 1,400°F) (US EPA, 1995b). The combustion of the sewage sludge is performed as the temperature is increased from 815°C to 925°C (1500°F to 1,700°F) (US EPA, 1995b). The gas residence times are usually four to five seconds (US EPA, 1995b). Emission controls can consist of wet scrubbers, wet electrostatic precipitators, afterburners, and regenerative thermal oxidizers (US EPA, 1995b). In a fluidized bed combustor, the sludge is typically combusted at temperatures ranging from 750°C to 925°C (1,400°F to 1,700°F) (US EPA, 1995b). The gas residence times are usually two to five seconds (US EPA, 1995b). Emission controls can consist of venturi scrubbers, multicyclones, fabric filters, activated carbon injection, and carbon bed absorbers (US EPA, 1995b).

SSI unit design and operation can vary widely across the nation. Current SSI standard operating conditions may not be effective for the treatment of PFAS. There is a concern with PFAS being emitted as products of incomplete combustion (PICs). A recent study performed on aqueous film forming foam (AFFF) showed that temperatures above 1100°C were necessary to promote PFAS destruction and minimize PICs (Shields et al., 2023). Additionally, longer residence times are recommended coupled with the use of high-temperature thermal oxidizers to reduce emissions. While this research was performed on a liquid-phase material and more research is still needed on semi-solid and solid-phase matrices, these findings indicate that current temperatures used for SSIs may not be high enough and the gas residence times may not be long enough to completely destroy PFAS. Furthermore, an additional recent study monitored PFAS fate from both a multi-hearth furnace and a fluidized bed combustor (Winchell et al., 2024). The PFAS testing found that the stack emissions from the multi-hearth furnace contained reportable levels of all targeted PFAS measured, representing an average of 5% of the total targeted PFAS monitored in the feed per sample run with emissions consisting mainly of shorter-chain PFAS (Winchell et al., 2024). Moreover, for both the multi-hearth furnace and fluidized bed combustor, nonpolar fluorinated organics were detected in the wet scrubber water streams, which were sourced from treated wastewater effluent (Winchell et al., 2024). Additional testing is still needed comparing more units of multi-hearth furnaces and fluidized bed combustors, while also using newly released air methods (*e.g.*, OTM-50) to test SSI emissions for more volatile PICs (US EPA, 2024f). Consequently, one issue is that volatile PFAS released as PICs may be inhaled by populations near the SSI and PICs could have the potential to transform and degrade into more persistent PFAS (*e.g.*, PFOA and PFOS), which can be distributed through atmospheric deposition to soil and water.

Due to these uncertainties around PFOA and PFOS destruction in SSIs, the potential for PICs to be released that degrade to PFOA and PFOS, and other uncertainties around thermal destruction conditions, the SSI model will not be quantitatively modeled for this draft risk assessment. However, the conceptual model in **Figure 7** illustrates which pathways may be available for PFOA and PFOS exposure after sewage sludge is incinerated. This conceptual model focuses on the deposition of PFOA and PFOS to soil or surface water bodies because of the absence of an inhalation hazard value for PFOA and PFOS. Once PFOA and PFOS are deposited on the soil surface or water surface, many of the same potential exposure pathways are available that were described in the prior conceptual models, including exposure through fish consumption, soil ingestion, food intake, and drinking water intake. The risk assessment will qualitatively discuss the potential for risk in these pathways in the SSI scenario. The EPA has provided information on incineration in its Interim Guidance on Destroying and Disposing of Certain PFAS and PFAS-Containing Materials That Are Not Consumer Products (US EPA, 2024g).



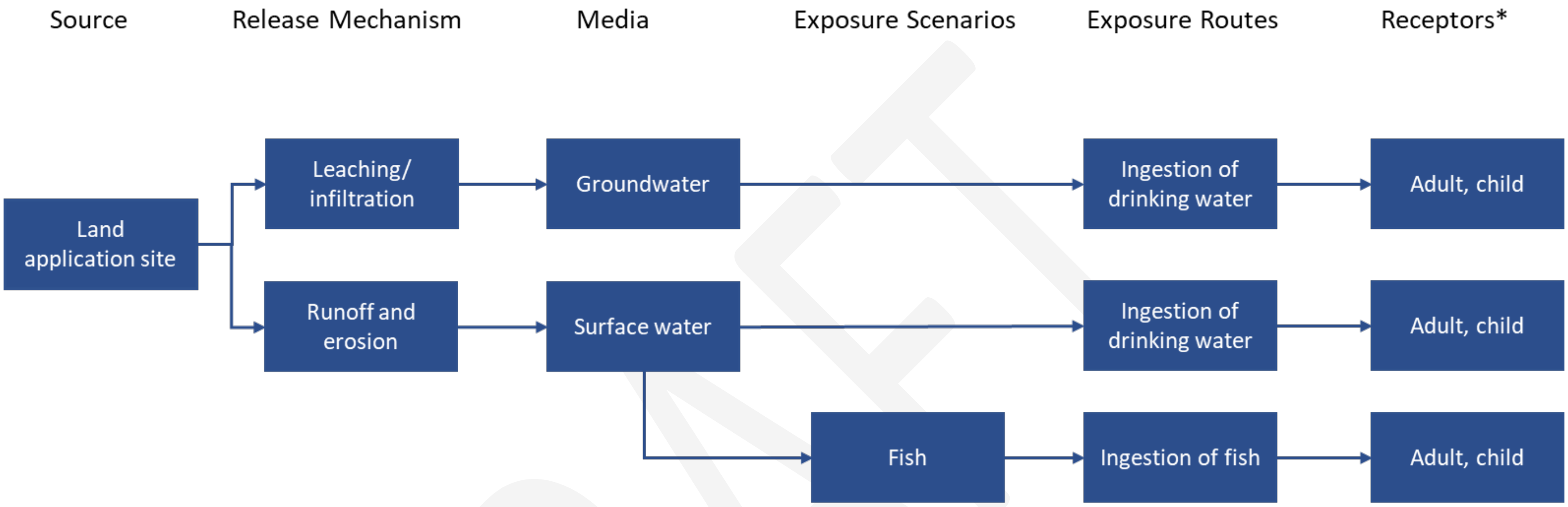
Note that deposition may also occur directly to exposed plants (forage or exposed produce), but this is typically a negligible pathway compared to soil uptake and is not shown.

* Receptor populations may include farm families, home gardeners, CSA participants, or nearby residents.

Figure 7. Conceptual model for sewage sludge incineration.

2.8.5 *Other Land Application Scenarios*

Biosolids land application can occur at many types of sites with low or high public contact, including forests, tree farms, road construction sites, golf courses, and more. A generic model for land application sites with low public contact would include potential pathways like leaching to groundwater and runoff to surface water, but it would not include pathways like ingestion of soil. The conceptual model in **Figure 8** illustrates which pathways may be available for PFOA and PFOS exposure in other land application scenarios.



* Receptor population is nearby residents.

Figure 8. Conceptual model for other land application scenarios.

The EPA has not modeled biosolids fate and transport in forest settings or other sites with low public contact in previous assessments (US EPA, 1995a; US EPA, 2003a). There is limited information available about the biosolids application rates in other types of land application related to roadway development, forestry, and others. Further, there are no pre-existing models for the fate and transport of biosolids applied to forests, tree farms, or other applicable sites, and studies to parameterize new models specific to biosolids fate and transport in silviculture are limited. Therefore, the EPA will assess these pathways qualitatively.

Biosolids that are applied at golf courses, parks, playgrounds, schools and homes may be Class A_{EQ}, meaning that the Class A pathogen requirements and the stricter requirements for chemicals in part 503 must both be met. Class A_{EQ} biosolids can be sold directly to the public, *e.g.*, at hardware stores, without any further requirements on the method, rate, or location of application. Restrictions that apply to other classes of biosolids that may reduce exposures do not apply to Class A_{EQ} biosolids. Since they also may be applied to residential locations, the potential for incidental soil ingestion by children becomes a more significant concern given the likelihood that a larger number of children may be repeatedly exposed at these types of sites.

The EPA's Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) program generates regional screening levels (RSLs) for residential soil for CERCLA hazardous substances based on the RfD for a chemical and a high-end incidental soil ingestion rate for children. PFOA and PFOS were added to the CERCLA hazardous substance list in May 2024, and the EPA developed screening values (1.9e-5 mg/kg PFOA, equivalent to 0.019 ppb; 6.3e-3 mg/kg PFOS, equivalent to 6.3 ppb, from US EPA 2024i), as starting points for determining if a chemical needs to be considered in a Superfund site's remediation plan. The incidental soil ingestion exposure pathway evaluated for CERCLA screening values is relevant to Class A_{EQ} biosolids that are land applied in places like parks, playgrounds, schools, and homes.

Finally, domestic septage is sometimes managed through land application to agricultural sites or other sites with low potential for human exposure (*i.e.*, turf farms, forested lands, and reclamation sites). Record keeping by the appliers is required for domestic septage land application for five years after the application, but these records are not required to be reported to the permitting authority. As a result, the EPA has limited data available on the types of lands used for domestic septage land application or the rates of application used. For more information on domestic septage application to grow crops, see the EPA's 2024 factsheets on Requirements for Application of Domestic Septage to Agricultural Land (US EPA, 2024j;k).

2.9 Analysis Plan

2.9.1 Modeling Plan

The CWA requires the EPA to ensure that the reuse of biosolids and disposal of sewage sludge does not adversely affect public health or the environment. To achieve this goal, the EPA conducts human health and ecological risk assessments for contaminants known or expected to be in biosolids (US EPA, 1993; 2003a; 2023c). In such risk assessments, the EPA conducts a series of modeling exercises with increasing refinement to estimate and characterize potential risks posed by activities associated with biosolids disposal or reuse (US EPA, 2023c). If the potential for risk exists from a chemical/contaminant in biosolids, the EPA typically determines the concentration of that chemical in biosolids that interfere with each use or disposal practice. The following sections describe the overall modeling approach that the EPA is taking for PFOA and PFOS in this draft risk assessment.

2.9.1.1 High End Deterministic

The EPA first screens chemicals that have been detected in biosolids using a high-end deterministic model for pasture and crop farming scenarios called the Biosolids Tool (BST; US EPA, 2023d). By using a high-end deterministic approach, the EPA ensures that its initial risk screening is conservative (health protective) in several ways. First, the screening tool uses modeling scenarios (crop and pasture farm) that generally result in the higher potential exposure rates than other biosolids reuse or disposal options. Second, the exposure modeling in this screening tool assumes high-end (95th percentile) consumption rates for drinking water, fish ingestion, produce consumption, and milk and meat consumption. Third, the exposure modeling assumes that the biosolids concentrations applied to the farm are at the 95th percentile of concentrations that have been measured in U.S. biosolids. The high-end deterministic model outputs estimated daily doses (mg/kg-day) that receptors are exposed to through each pathway in the model. These estimated exposures are then assessed individually against the available toxicity values. EPA used the BST to screen PFOA and PFOS, finding that every pathway modeled indicated that this scenario could result in excess risk (a summary of the BST inputs and screening results for PFOA and PFOS can be found in Appendix E). These findings motivated EPA to further assess the fate and transport of these compounds in various biosolids use and disposal scenarios. This high-end deterministic assessment approach is similar to the approach used in prior sewage sludge risk assessments (US EPA, 1992; 1995a) which focused on identifying risks to someone with a "reasonable maximum exposure."

2.9.1.2 Central Tendency Deterministic

Given the results of high-end deterministic modeling for PFOA and PFOS in the farming scenarios (see Appendix E), the EPA decided to assess risks under median conditions rather than high-end conditions, to better understand the potential scope and magnitude of potential risks under different use and disposal scenarios. Given that all sewage sludge requires some type of disposal or reuse management activity, it is also important to understand risks from biosolids used in the farm (crop, pasture) scenarios in the context of other use and disposal scenarios, such as land reclamation, silviculture, surface disposal or incineration. Completing a central tendency deterministic modeling exercise for multiple reuse and disposal options provides an understanding of exposure risks associated with biosolids at conditions that approximate average conditions for each use scenario. This intermediate step between high-end deterministic screening and refined probabilistic risk assessment can help inform which scenarios, if any, should be the focus of more refined risk modeling (*i.e.*, deriving risk-based values protective of the 95th percentile exposure scenario using Monte Carlo analysis).

To complete the central tendency deterministic modeling steps, the EPA 1) assessed available fate and transport models to ensure that they are the best available models for assessing PFOA and PFOS and 2) parameterized the modeling inputs to reflect an overall set of median U.S. conditions. Section 2.9.2 discusses the model selection process for refining the PFOA and PFOS fate and transport modeling. Section 2.9.3 discusses the input parameters used for this modeling approach. At a high level, the input parameters for this central tendency deterministic modeling exercise represent less health protective assumptions than the EPA would typically use in a risk assessment for biosolids or other environmental media. For example, the EPA is assuming that the drinking water intake rate is about 1 L/day for an 80-kg adult, compared to the 90th percentile value of 2.4 L/day that is typically used for CWA purposes.

2.9.1.3 Probabilistic (Monte Carlo Analysis)

Monte Carlo simulation is a statistical technique by which a quantity is calculated repeatedly, using randomly selected values from assigned distributions for each calculation. These results approximate the range of possible outcomes and the likelihood of each. When Monte Carlo simulation is applied to risk assessment, risk appears as a frequency distribution rather than a single value, which allows for the identification of risks at specific percentiles. Previous sewage sludge risk assessment has used a Monte Carlo probabilistic modeling approach and targeted risk-based thresholds protective of 95% of the modeled population (US EPA 2003a). This draft risk assessment does not include Monte Carlo Analysis because the central tendency deterministic modeling indicates that risks are prevalent even when targeting median (50th percentile) conditions in individual exposure pathways. See section 4.9 for more discussion on the EPA's rationale for not conducting Monte Carlo Analysis in this assessment.

2.9.2 Model Selection

This draft risk assessment relies on several independent models to understand PFOA and PFOS fate and transport across the exposure scenarios (crop farm, pasture farm, reclamation site, and surface disposal site). In the farming and reclamation scenarios, the first step is to model how much PFOA and PFOS sorb to soil, are moved through runoff and erosion, and move through the unsaturated zone and saturated zone into groundwater after biosolids have been land-applied to soils. A separate model estimates the amount of PFOA and PFOS moving through runoff and erosion; this model then estimates the concentrations of PFOA and PFOS that enter a nearby lake or reservoir. Finally, a third model estimates the amount of PFOA and PFOS moving through groundwater to nearby drinking water wells. In the surface disposal scenario, a model is used to estimate how much PFOA and PFOS may leach through the underlying soil from a lined or unlined surface disposal site. Then the same groundwater model used in the farming and reclamation scenarios is used to understand how leached PFOA and PFOS move through groundwater to neighboring groundwater wells. The following sections describe how and why EPA selected various models for this assessment.

The results of the fate and transport modeling include concentrations of PFOA and PFOS in environmental media over time, such as soil concentrations on the farm field, soil concentrations on nearby "buffer" land, surface water concentrations in the nearby lake or reservoir, and groundwater concentrations at wells with given depths and distances from the field. These media concentration results are then used to calculate concentrations of PFOA and PFOS in drinking water, vegetables, fruits, feed crops, livestock products (milk, beef, chicken, eggs), and fish using various uptake factors, such as biotransfer factors (BTFs) and bioaccumulation factors (BAFs). Finally, the concentrations of PFOA and PFOS in each media type are used to calculate exposure and risk to the relevant receptors in each conceptual modeling scenario.

2.9.2.1 PFOA- and PFOS-specific Fate and Transport Considerations

The mobility of PFOA and PFOS in the environment, an active area of research, is known to be affected by a number of factors, including:

- hydrophobic/hydrophilic-surfactant behavior (*e.g.*, fluid-fluid or air-fluid interface retention);
- attraction to the solid phase in sediment (Higgins and Luthy, 2006), sludge (Milinovic et al., 2016), soil (Milinovic et al., 2015), and organic carbon in general (Higgins and Luthy, 2006);
- ionic behavior as a function of pH (Place and Field, 2012; Pereira et al. 2018); and
- competition among these processes.

Methodologies for assessing the impact of PFOA and PFOS retention at the air-water interface (AWI) have been proposed (Brusseau, 2018), modeled (Guelfo et al., 2020), and implemented in various fate and transport simulators (Guo et al., 2020, 2022; Silva et al., 2022). The fact that AWI has been shown to be a significant factor in PFAS fate and transport has focused modeling efforts on the vadose zone though AWI retention is also relevant to saturated aquifer environments because some air may be entrained in pore spaces of the saturated zone (Bumb et al., 1992). In equilibrium transport modeling, it is assumed that sorption occurs at much faster rates than the residence time of groundwater. However, studies have observed that solid phase sorption processes for PFOA and PFOS are not always well represented by reversable equilibrium partitioning assumptions due to rate-limited air-water interfacial adsorption and fluid-fluid interfacial adsorption (Guelfo et al., 2020; Brusseau, 2020). Sorption of PFOA and PFOS to non-advective domains influences the magnitude and timing of transport from the vadose zone to groundwater.

Soil transport modeling studies that incorporate PFOA- and PFOS-specific, non-linear adsorption processes predict that even after the source of PFOA and PFOS in the vadose zone has been discontinued, PFOA and PFOS mass can remain in the vadose zone for decades, centuries, or longer (Zheng & Guo, 2023). In some of this soil modeling, PFOA and PFOS appear to not break through the vadose zone and enter groundwater aquifer for hundreds or thousands of years after they are applied to the surface (see Section 3.2.3 and Zheng & Guo, 2023). Given that PFOA and PFOS manufacturing only began in the 1940's, this modeling would suggest that groundwater contamination associated with land application of PFOA and PFOS contaminated biosolids would not be observed for many years into the future. However, instances of high groundwater concentrations of PFOA and PFOS have been documented for both shallow and deep vadose zones (Brusseau et al., 2020; Dauchy et al., 2019) and in various states including Maine, Michigan, and Alabama, where PFOA and PFOS contamination is attributable to land-applied biosolids (see Section 6). For example, Brusseau et al. 2020, in their review of PFAS concentrations at contaminated and non-contaminated soil sites, found that though PFOS concentrations are highest in the upper portion of the soil profile (as expected), PFOS is still present at significant (~1-10 ppb) concentrations in soil samples from 25-40 m below the surface. If PFOA and PFOS were so successfully retained in the surface soils and upper vadose zone subsurface soils, these real-world examples of transport deep in the vadose zone and groundwater would not be expected.

Soil heterogeneities, preferential transport pathways, and colloidal transport mechanisms are environmental characteristics that are often omitted from modeling studies and that may be responsible for faster migration of PFAS through the vadose zone than is expected from current modeling (Zeng and Guo, 2021; Bierbaum et al. 2023). These factors may also result in more PFOA and PFOS mass being transported through the soil column than is estimated using currently available models, resulting in higher observed groundwater concentrations.

The EPA assessed fate and transport models that explicitly include retention on the AWI (such as the Guo et al. 2022 model) and existing EPA models that can be parameterized to better reflect PFOA and PFOS transport behavior (see appendix C). However, the ability of any model to reliably predict the timing of PFOA and PFOS impacts to groundwater in highly characterized, non-idealized environments is low (Zeng and Guo, 2021). In fact, available models (Guo et al. 2022 and EPA models) likely overestimate the time required for PFOA and PFOS to reach groundwater, and this modeling of PFAS transport in soil systems remains an active area of ongoing research. Consistent with previous sewage sludge risk assessments, this draft risk assessment will consider the peak groundwater concentrations when calculating risks, regardless of the timing of their occurrence, to avoid underestimating risks through this pathway (US EPA, 1992; US EPA 2003a;b).

In addition to transport models for PFOA and PFOS movement through soil, this draft risk assessment also requires models for understanding transport to surface waters and groundwater transport. The following sections describe which models were assessed and selected for use in this assessment.

2.9.2.2 Soil Surface Modeling

The underlying model for the evaluation of the concentrations of PFOA and PFOS in soil is the EPA's Multimedia, Multipathway, and Multireceptor Risk Assessment (3MRA) modeling system (US EPA, 2003f;g), developed by the EPA Office of Land and Emergency Management. The 3MRA modeling system includes a number of modules.

The Land Application Unit (LAU) module within 3MRA models the incorporation of contaminants in biosolids into the top layer of soil and then simulates:

- The vertical movement of those contaminants through the top 20 cm of soil, estimating a leachate mass flux that is used by the EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP) to model transport through the vadose zone to groundwater;
- The horizontal movement of those contaminants via erosion and runoff from the field to a buffer area, and ultimately to the surface water body, estimating a waterbody load that is used by a surface water model, the Variable Volume Waterbody Model (VVWM), to model transport within the waterbody; and
- The losses of contaminant to air via wind erosion of particulates; this mass is removed from the LAU but is not modeled further.

The mass that remains after these processes is the basis of the soil concentration in the top layer of soil that is available for plant uptake, soil consumption by livestock, and consumption of soil by humans.

The 3MRA model has been peer reviewed and used extensively to support regulatory risk assessments conducted for EPA's Office of Resource Conservation and Recovery and Office of Water (US EPA, 2003f). As part of the 3MRA modeling system, the LAU source module was developed to estimate annual average surface soil constituent concentrations and constituent mass emission rates to air, downslope land, and groundwater. These estimates are passed to other environmental fate and transport models and used to calculate exposure and risk. Additionally, the LAU source module incorporates a local watershed submodule (a "local" watershed is a sheet-flow-only watershed containing the LAU and a downslope buffer area between the LAU and the waterbody) to provide estimates of constituent mass flux rates from runoff and erosion from the field to the downslope buffer, and then from the buffer to a downslope water body (called the drinking water reservoir in conceptual models for the pasture and crop farm). The LAU module also produces constituent soil concentrations on the field, as well as in the downslope buffer area.

The LAU model conserves mass while accounting for releases from the agricultural field via leaching, volatilization, particulate emissions, runoff, and erosion, and release from the buffer via runoff and erosion to the waterbody. The model also accounts for deposition onto the plants on the field, but not back onto the soil of the agricultural field or buffer, so soil concentrations in these areas may be slightly underestimated. Though the LAU model can account for abiotic degradation, biodegradation, and volatilization, these factors are not relevant for PFOA and PFOS (ATSDR, 2021).

The specific inputs and the data used in the LAU source model are presented in Appendix B. The LAU model runs for 150 years, starting with the year of first application, and outputs a time series of daily and annual average soil concentrations for the field and the buffer, daily and annual average concentrations of contaminant mass, eroded solids, and runoff from the field and buffer, annual average leachate concentrations, and air emission rates (particulate). This assessment assumes that land application occurs for 40 years and then stops. Peak concentrations in the soils, runoff, and leachate are expected to occur around the time application ceases, however, the longer simulation time allows for confirmation that 150 years is sufficient to capture peak concentrations in these media.

The 3MRA Surface Impoundment module is used to model the amount of PFOA and PFOS that may be released from a surface disposal site to the vadose zone under the site. The resulting leachate mass flux is used by EPACMTP to estimate transport through the vadose zone and groundwater. The inputs used to parameterize the surface impoundment model are also included in Appendix B.

2.9.2.3 Surface Water Modeling

Erosion and runoff loadings from the downslope buffer area (calculated by the 3MRA LAU module) are fed into VVWM (US EPA, 2019b), developed by the EPA Office of Pesticide Programs for estimating concentrations in surface water bodies. The VVWM model estimates concentrations of PFOA and PFOS in a drinking water reservoir; dissolved concentrations in the water column are used to calculate risks associated with drinking water whereas total water column concentrations are used to calculate fish tissue concentrations using BAFs.

VVWM was developed from another EPA model, the Exposure Analysis Modeling System (EXAMS; Burns, 2000) that simulates standard water bodies that receive chemicals from the standard field. VVWM behaves much like EXAMS, simulating the US EPA standard water bodies (*i.e.*, farm pond and index reservoir), but with greater efficiency and flexibility. The VVWM also allows for variations in water body volume daily due to runoff, precipitation, and evaporation. Temperature, wind speeds, and chemical dissipation processes are also allowed to vary daily.

The VVWM consists of two regions: a water column and a benthic region (US EPA, 2019b). Each individual region is completely mixed and at equilibrium with all phases in that region, with equilibrium described by a linear isotherm. The two regions are coupled by a turbulent-mixing, first-order mass-transfer process. The water volume may vary by inputs of precipitation and runoff and by outputs of evaporation and overflow. Degradation via biodegradation, hydrolysis, and photolysis can be parameterized for each compartment as applicable in VVWM, but PFOA and PFOS do not undergo these degradation processes (ATSDR, 2021).

2.9.2.4 Groundwater Modeling

Modeling of the groundwater pathway is accomplished using two models: a model responsible for releasing PFOA or PFOS into the subsurface, the 3MRA LAU Module, and a subsurface flow and transport model, EPACMTP (US EPA, 2003d;e). The 3MRA source modules calculate the amount of PFOA and PFOS that leave the top layer of soil for the LAU or the bottom of the surface disposal unit as part of the leachate. The maximum mass flux of any constituent in the modeled leachate occurs in all cases during the application period and is fully captured within the 150-year modeling timeframe of the source modules. The subsurface model (EPACMTP) is allowed to run as long as 10,000 years if necessary to observe the peak groundwater concentrations at simulated wells (see Section 2.9.2.1 for discussion of the known modeling deficiencies in predicting the timing of groundwater impacts and see Section 3.2.3 for a discussion of modeled groundwater concentrations over time).

EPACMTP is then used to calculate the amount of PFOA and PFOS that travel through the remaining soil column (the vadose zone) to the groundwater table and downgradient to a drinking water well located 5 meters from the edge of the field or surface disposal unit (in the middle of the 10-meter-wide buffer). The modeled depth of the vadose zone varies depending on the geographical location. As described in Section 2.9.2.1, PFOA and PFOS present challenges for calculating soil transport compared to typical organic contaminants due to their surfactant properties. PFOA and PFOS can reside at the air-water interface and electrostatically sorb to minerals in soils after moving into the vadose zone. Depending on the hydrogeology and minerology of the location, this may retard the movement of the chemical into the groundwater table. EPACMTP has been used within the EPA for decades to estimate subsurface transport through the vadose zone to groundwater but has not traditionally been parametrized to estimate air-water interface effects.

The hypothetical drinking water well in EPACMTP is represented by four observation locations placed at 0.5, 1.0, 1.5, and 2.0 meters below the water table to ensure the maximum groundwater concentration is observed. The highest concentration observed across the four depths is used to calculate a proportional constant that represents the minimum cumulative reduction and attenuation of leachate concentrations as they migrate through the subsurface to the drinking water well.

In Appendix C (groundwater modeling), models besides EPACMTP are compared for their relevance to PFOA and PFOS vertical transport through the vadose zone. Other models can incorporate PFAS-specific parameters like air-water interface effects and nonlinear adsorption. These factors result in lower peak groundwater concentrations and longer delays in the transport of PFOA and PFOS to the groundwater at the farm. EPACMTP estimates arrival times of aquifer contamination at the water table that are, in some cases, much longer than those that have been observed at biosolids application sites in Maine and Michigan, but closer to those observed breakthrough times than models that incorporate air-water interface effects and nonlinear adsorption. For this reason, EPACMTP was selected as being more appropriate for modeling vertical transport through the soil column.

The model implementation also includes some assumptions to protect groundwater resources now and in the future. Firstly, the draft risk assessment assumes that drinking water receptors have wells that are placed in the center of the buffer, five meters from the edge of the field or surface disposal unit and centered around the highest concentration in the groundwater plume below the water table. If a homeowner had a deeper well or a well located on the fringe of the plume, rather than the center of the plume, they would have lower drinking water concentrations and lower risks. The draft risk assessment also presents exposures that occur during the years with the highest media concentrations for soils, surface water, and groundwater. While these assumptions may overestimate risk to a specific person at a specific site, they are reasonable for the purpose of a national draft risk assessment seeking to determine if levels of PFOA and PFOS in sewage sludge may adversely affect human health or the environment. For example, it is important to protect the groundwater as a source for potential drinking water regardless of when that peak may be reached or where a well may be placed.

2.9.2.5 Air Dispersion Modeling

Generally, the EPA uses AERMOD to parameterize the transport of most chemicals from farm fields; however, for the PFOA and PFOS assessment, the volatilization rate has been set to zero and no dispersion modeling is needed. The only airborne loss of PFOA and PFOS is due to wind erosion emissions of dust from the field, and this loss is calculated within the 3MRA LAU module.

2.9.2.6 Plant and Animal Uptake Equations

The produce, meat, and milk exposures are calculated using the methodology found in the Human Health Risk Assessment Protocol (HHRAP; US EPA, 2005b), developed for hazardous waste combustion facilities and slightly modified to account for the available data for parameterization. The fate and transport models generate the estimated concentrations of the contaminated media that are used to calculate concentrations in crops and animal feed (Equation 1), animal products (Equation 2), and fish tissue (Equation 3).

Crops

Equation 1. Crop Concentrations Due to Root Uptake from Soil, $P_{produce}$, P_{feed} (mg/kg)		
Produce (Aboveground Fruits and Vegetables, Root Vegetables)		***Feed crops (Forage and Silage)***
$P_{produce} = C_{soil} \times B_r \times \left(\frac{100 - MAF}{100}\right)$		$P_{feed} = C_{soil} \times B_r$
Name	**Description**	**Source**
$P_{produce}$, P_{feed}	Concentration of contaminant in crops (aboveground fruits or vegetables, and root vegetables or animal feed (P_{feed})	Calculated
C_{soil}	Concentration of contaminant in soil, averaged over tilling depth (mg/kg)	LAU model output
Br	Soil-to-plant bioconcentration factor: $\frac{[plant](\frac{mg}{kg} dry\ weight)}{[soil](\frac{mg}{kg} wet\ weight)}$	See model parameterization, Section 2.9.3.4
MAF	Plant tissue-specific moisture adjustment factor to convert dry weight concentrations into wet weight (percent)	See model parameterization, Section 2.9.3.4
100	Conversion factor from percent to fraction (unitless)	NA

Livestock

Equation 2. Concentration in Animal Products, A (mg/kg WW)		
where	$A = BTF \times [I_{soil} + I_{feed} + I_{water}]$ $I_{soil} = C_{soil} \times Q_{soil} \times Bs$ $I_{feed} = \sum_i P_i \times Q_i \times F_i$ $I_{water} = C_w \times Q_w \times F_w$	
Name	**Description**	**Value**
A	Concentration in the animal product (beef, chicken, egg, milk)	Calculated
I	Livestock intake of soil (I_{soil}), feed (I_{feed}), and water (I_{water})	Calculated
BTF	Biotransfer factor for animal product: $\frac{[animal\ product](\frac{mg}{kg} ww)}{intake\ rate(\frac{mg}{day})}$	See Section 2.9.3.5
C_{soil}	Average concentration in surficial soil (mg/kg)	LAU model output
Q_{soil}	Quantity of soil consumed per day (kg/day)	See Section 2.9.3.6
Bs	Bioavailability factor in soil (fraction)	1
P_i	Average concentration in plant type *I* (forage, silage, grain) (mg/kg DW)	Calculated; see Equation 1 for forage and silage. Grain assumed to be purchased from an uncontaminated source
Q_i	Quantity of plant type *I* consumed per day (kg DW/day)	See Section 2.9.3.6
F_i	Fraction of plant type *i* grown in contaminated soil (unitless)	See Section 2.9.3.6
C_{gw}	Average concentration in groundwater (mg/L)	LAU model output
Q_w	Quantity of water consumed per day (L/day)	See Section 2.9.3.6
F_w	Fraction of water contaminated (unitless)	1

Fish

Equation 3. Concentration in Fish Filet, C_{filet} (mg/kg)		
$C_{filet} = C_{tot} \times BAF$		
Name	**Description**	**Value**
C_{dtot}	Total water column concentration (mg/L)	VVWM model output
BAF	Bioaccumulation factor for fish filet (L/kg)	See Section 2.9.3.7

2.9.3 Model Parameterization

To calculate exposure and risk for the pathways depicted in the four conceptual modeling scenarios where the EPA is quantitatively assessing outcomes (crop farm, pasture farm, surface disposal site, and land reclamation), the EPA must parameterize hundreds of values used in fate and transport and exposure models. This includes parameters related to the fate and transport of PFOA and PFOS in soil columns, groundwater, surface water, and into crops and animals. These parameters also include toxicity values for PFOA and PFOS and exposure factors for the many pathways of human exposure depicted in the conceptual models. Finally, these parameters include characteristics of the modeled environment, like the size of the modeled surface water reservoir or the size of the field receiving biosolids for land application.

Establishing chemical-specific values for some of these parameters can be challenging for PFOA and PFOS because these chemicals present different characteristics than are typical for other organic chemicals. For example, while some environmental fate parameters for other organic compounds can be predicted using the water-octanol partitional coefficient (K_{ow}), this value cannot be measured for PFOA and PFOS because of their surfactant properties; experimental data are needed for these parameters instead. If there is an existing assessment available from the EPA or another agency that is relevant to a chemical-specific parameter, the conclusions of those assessments are prioritized over results from individual studies. For example, rather than compile individual human health toxicity studies for PFOA and PFOS, this draft risk assessment will rely on the conclusions of the EPA's Final Toxicity Assessments for PFOA and PFOS (US EPA 2024b;c) as a source of toxicity values (reference doses and cancer slope factors). Similarly, this draft risk assessment will rely on fish uptake factors (bioaccumulation factors) presented in EPA's Draft Human Health Criteria for PFOA and PFOS (US EPA, 2024o;p).

For many of the fate, transport, and uptake parameters, there are no relevant existing assessments that can be used for parameterizing model inputs needed for this assessment. In these cases, the EPA searched and reviewed the available peer-review literature. The EPA applied the following hierarchy to the available fate and transport studies:

1. Field studies from sites with biosolids application
2. Laboratory (including greenhouse) studies using biosolids-amended soils
3. Field studies from other types of PFAS-impacted sites
4. Laboratory (including greenhouse) studies using materials with other sources of PFAS contamination.

Study quality metrics relevant to each study type are described in this section. When there are multiple measurements or studies of sufficient quality available for the same parameter in the same data tier, the EPA used the median value to parameterize the models. As an example, plant uptake factors were determined by prioritizing studies where biosolids contaminated with PFOA and PFOS were applied in the study area/field. If there were multiple acceptable field-studies available where the source of PFAS contamination was sewage sludge, the median of these data was selected for the study parameter.

While field studies are generally preferred over laboratory studies for most parameters, field studies with real-world contamination are likely to include potential confounders, including other PFAS, which may or may not be precursors to PFOA and PFOS. While use of these studies may overestimate PFOA and PFOS transport or uptake in some settings, the degree of PFAS diversity seen in these real-world field studies is not dissimilar to the degree of PFAS diversity found in biosolids (Thompson et al., 2023a). For this reason, the benefit of using biosolids-specific data in most cases outweighs the uncertainty contributed from the potential for PFOA and PFOS precursors or other confounders to influence the final parameter values.

Several conceptual models are based on agricultural sites, where a farm family's exposure is modeled. When parameterizing human exposure factors, food consumption data specific to home-produced foods or consumption rate amongst farmers are prioritized over general population data. This draft risk assessment uses exposure factors presented in the most recently updated version of the EPA's Exposure Factors Handbook (EFH; US EPA, 2011) chapter for home-produced foods (Chapter 13), when available. If there are not data specific to home-produced food available, chapters of the EFH describing the general population are used. Some of these chapters have been updated since 2011 and issued as separate documents; in all cases, the most recent update is used and referenced.

The environmental fate and transport models used in this draft risk assessment also require parameters related to the environmental setting, such as size of the field used for land application. When these parameters are not specific to sewage sludge use and disposal (for example, porosity of benthic sediments in the waterbody near to the field), default values provided in the peer-reviewed EPA model or values from previous EPA sewage sludge assessments are used (US EPA, 2003a). When these parameters are specific to a setting, regionally representative values from a wet, moderate, and dry climate in the US are used (see Section 2.9.3.13). When the parameters are relevant to practices for sewage sludge land application or disposal, median values from relevant US datasets are used, consistent with prior sewage sludge risk assessments (US EPA, 1992; 1995a; 2003a). Descriptions of the selected values for each parameter are included below. Tables of values used for each parameter are also summarized in Appendix B.

2.9.3.1 Toxicity Values

In 2024, the EPA published final human health toxicity assessments for PFOA and PFOS (US EPA 2024b;c). These final assessments include an RfD and CSF for PFOA and PFOS. These values are relevant to all oral ingestion pathways, including drinking water and dietary intake. While PFOA and PFOS exposures have been associated with numerous adverse health outcomes in humans, the RfDs and CSFs are derived based on the most sensitive adverse health outcomes; protecting against these outcomes will also protect against the outcomes that occur after higher levels of exposure. PFOA and PFOS are classified as likely carcinogens (L). The biosolids exposure models assesses the cancer risks and non-cancer risks associated with each exposure pathway. As described in Section 2.6.1, the RfDs and CSFs for PFOA and PFOS are as follows:

Table 6. Toxicity Values for PFOA

Toxicity Value Type	Value	Critical Effect(s), Critical Study/Studies
RfD (based on epidemiological data)	3 x 10^{-8} mg/kg/day	Reduced antibody response to vaccinations in children (diphtheria and tetanus) (Budtz-Jorgensen & Grandjean, 2018); decreased birth weight (Wikstrom et al., 2019); increased serum total cholesterol (Dong et al., 2019)
CSF (based on epidemiological data)	29,300 $(mg/kg/day)^{-1}$	Renal cell carcinoma (RCC) (Shearer et al., 2021)

Table 7. Toxicity Values for PFOS

Toxicity Value Type	Value	Critical Effect(s), Critical Study/Studies
RfD (based on epidemiological data)	1 x 10^{-7} mg/kg/day	Decreased birth weight (Wikstrom et al., 2019); increased serum total cholesterol (Dong et al., 2019)
CSF (based on animal toxicological data)	39.5 $(mg/kg/day)^{-1}$	Combined hepatocellular adenomas and carcinomas in female rats (Thomford, 2002; Butenhoff et al., 2012, 1276144)

2.9.3.2 Sewage Sludge PFOA, PFOS Concentration and Other Characteristics

The fate and transport models used in this assessment require a starting concentration for PFOA and PFOS in sewage sludge. For this central tendency assessment, the EPA is using low starting concentrations of 1 ppb for PFOA and 1 ppb for PFOS (dry weight). These values were selected because they represent a concentration that is lower than most sewage sludge generated in the U.S., including sewage sludge that represents only domestic sources (see Section 2.4 and Appendix A). This value is also near the reporting limits expected in most major laboratories using EPA Method 1633 on sewage sludge (US EPA 2024d). Notably, the models and calculations used in this draft risk assessment result in a linear relationship between the starting concentration of PFOA or PFOS in sewage sludge and the resulting concentrations and risks. For example, if modeling a starting concentration of 1 ppb PFOA results in a hazard quotient of 1 or a cancer risk level of 1 in 1 million (1 x 10^{-6}), a starting concentration of 10 ppb PFOA would result in a hazard quotient of 10 and a cancer risk level of one in one hundred thousand (1 x 10^{-5}). Should the EPA's draft modeling find risks in a given potential pathway with this low starting concentration of PFOA and PFOS in sewage sludge, it is reasonably anticipated that these risks could be prevalent across use and disposal sites in the U.S.

The fate and transport models also require additional characterization of the sewage sludge, including the dry bulk density, the fraction organic carbon, and the silt content of the sewage sludge. The silt content was assumed to be 10% (the median of the distribution provided in the 2003 risk assessment documentation). The bulk dry density of biosolids was assumed to be 0.7 g/cm^3, which is the median of the range provided in Gunn et al. 2004. This value was updated from the 2003 risk assessment documentation, which reported a bulk dry density of 1.6 g/cm^3, cited from the Technical Support Document for the Land Application of Sewage Sludge (US EPA, 1992).

2.9.3.3 Physical and Chemical Properties

PFOA and PFOS partitioning data between water and soil are needed to model the fate and transport of these chemicals through the environment. To represent solid-phase sorption of PFOA and PFOS in environmental media potentially affected by land-applied biosolids, the modeling framework uses the organic carbon distribution coefficient (K_{oc}). K_{oc} is then used to calculate the solid-phase adsorption coefficient (K_d) using the fraction of organic carbon (f_{oc}) in each modeled soil type. The EPA conducted a literature search to aggregate measured K_{oc} and K_d values for PFOA and PFOS in biosolids field studies, other field studies, and laboratory studies. The methodology and results from this literature search are described in Appendix C, Section C.3.2.1.

Based on this literature review, the EPA concluded that there are a range of K_{oc} values reported under various environmental conditions in soils. To represent the range of potentially relevant K_{oc} values at each site, the EPA modeled a "low-K_{oc}" scenario and a "high-K_{oc}" scenario, representing the 10th and 90th percentiles of the distribution, respectively. The values for PFOA and PFOS are provided in Tables 8 and 9, respectively. See Appendix C for more information on the distribution of observed K_{oc} values for PFOA and PFOS.

Table 8. K_{oc} Values for PFOA

Scenario	Value	Source
Low K_{oc} (10th percentile)	26 cm^3/g	Literature search; see text and Appendix C
High K_{oc} (90th percentile)	1,100 cm^3/g	Literature search; see text and Appendix C

Table 9. K_{oc} Values for PFOS

Parameter	Value	Source
Low K_{oc} (10th percentile)	250 cm^3/g	Literature search; see text and Appendix C
High K_{oc} (90th percentile)	22,000 cm^3/g	Literature search; see text and Appendix C

K_{oc} values are used in the model to estimate K_d values in four media: surface soil, subsurface soil, sediment, and suspended sediments. The sediment and suspended sediment values are for the drinking water reservoir. The f_{oc} is multiplied by the K_{oc} to obtain the K_d for each medium.

As described above, K_{oc} values vary for PFOA and PFOS in different studies and K_d will vary across sites. Other soil parameters including protein content and oxalate-extractable iron and aluminum may also be more relevant for a particular site for soil adsorption and K_d. Oxalate-extractable iron and aluminum content may be particularly relevant to deep soil settings, where organic matter content is low. The goal of this modeling activity is to provide estimates of a range of transport behaviors as this parameter is varied to reflect the environmental variability that will occur at different locations. To achieve this the model has been parameterized with multiple K_{oc} values and to calculate K_d across the simulated media in the 3MRA model including soil, biosolids, and sediments in the drinking water reservoir. Table 10 shows the values used to represent f_{oc} across each type of media.

Table 10. Fraction Organic Carbon Values by Medium

Medium	f_{oc} Value	Reference
Natural soil under the field	0.0118	STATSGO (USDA, 1994)
Bed sediments	0.04	VVWM (US EPA, 2019b)
Suspended sediments	0.04	VVWM (US EPA, 2019b)
Biosolids	0.40	Biosolids 2003 (US EPA, 2003a)

The LAU model uses both the biosolids f_{oc} and the soil f_{oc} and calculates a depth-weighted average of the two over the total incorporation depth (20 cm for crop, 2 cm for pasture or reclamation). For the purposes of this average, the waste depth is the application rate for a single application divided by the biosolids bulk density, and the soil depth is the rest of the application depth. The result is a higher f_{oc} than the soil value, but lower than the biosolids f_{oc}.

Both PFOA and PFOS are stable in air and water (UNEP, 2015; ATSDR, 2021), so half lives in air and water were not used. Other chemical-specific property values may be found in Appendix B.

2.9.3.4 Plant Uptake Factors

Bioconcentration factors (BCFs) are the uptake factors used for plants and are defined as the concentration of the compound in the relevant compartment of the plant divided by the concentration of the compound in the underlying soil. BCFs are unitless. Plant BCFs can be derived from studies with various experimental designs as long as the study measures concentrations of the chemical in the plant tissues and the soil. Surveys of plant tissue concentrations alone (*e.g.*, market surveys) are not useful for modeling or generating plant BCFs as they lack the corresponding soil data, though these studies can be used for general context of what types of exposures may be occurring. Because the matrix of biosolids and natural field conditions may impact the accumulation of PFOA and PFOS into plants, the following data hierarchy for plant BCF studies is used in this assessment:

1. Field studies with biosolids-amended soil
2. Greenhouse studies of potted plants with biosolids-amended soil
3. Field studies with other sources of PFAS contamination impacting the soil

This data hierarchy allows the EPA to preferentially select studies with biosolids-specific contamination sources and field conditions over other types of studies, as data are available.

The following literature search strategy was used to identify potentially relevant studies:

Database searched: PubMed

Search string: Title/Abstract search, (“PFAA”OR “PFAS” OR “PFCA” OR “PFOA” OR “PFOS”) AND (“food” OR “crop*”) AND (“soil” OR “biosolid*” OR “sludge”)

Date searched: 3/15/2024. No date limitations on results.

Relevant federal and state government reports are also included.

Results: 133 studies and results from recent literature surveys by Lesmester, 2023 and Li, 2022

The following criteria must be met:

- Measured PFOA or PFOS concentrations in plants and soil
- Study must relate to one of the 3 categories in the data hierarchy
- Known source of contamination
- Soil not contaminated by spiking (lab contaminated soil)

Several key findings of the following papers include that grass and leafy greens likely exhibit the highest soil to plant uptake (or plant BCF) values amongst the plants that have been studied. Roots and tubers that are consumed (*e.g.*, carrots) may also have high uptake, but a field study does not exist to verify the available greenhouse data (Wen et al., 2016) for that compartment of plants.

For PFOA and PFOS, fruits and seeds have lower uptake than the stems and leaves (vegetative parts of the plants), likely due to the need to cross additional membranes to reach the fruit and seeds.[11] Blaine et al., 2013 collected corn stover (stalk, leaves, and cobs), corn grain, and soil from biosolids amended fields in the Midwest; these researchers found no detectable PFOA or PFOS in corn grain from fields with soil concentrations of 4.4 ng/g PFOA and 4.3 ng/g PFOS. Blaine et al., 2014, a greenhouse-based study

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[11] Note that pea pods, tomatoes, and eggplants are fruits like apples, oranges, and blueberries are the ripened ovary of a plant and therefore “fruits” from a botanical perspective. However, some dietary surveys may create confusion based on common usage of terms like vegetable and fruit.

of plant uptake using biosolids-amended soils, similarly found that uptake into fruit was one to three orders of magnitude less than uptake into roots or shoots for PFOA and PFOS.

Authors of most studies have estimated uptake factors based on dry weight concentrations in the plant matter. The basis of the soil concentration is reported in most of the following articles and specified as dry weight. The model calculates and applies these uptake factors to wet weight of soil; therefore, while the study discussions below present the data as reported (dry weight plant concentration to dry weight soil concentration), the final BCFs presented in Appendix B have been converted to dry weight plant concentration to wet weight soil concentration using field capacity (water content of soil) and porosity (water plus air content) of soil for feed crops; no further conversion is necessary as animal dietary data are also commonly expressed on a dry weight basis. For fruits and vegetables consumed by children and adults, the relevant consumption data are available on a wet weight basis, so a moisture adjustment factor (MAF) is needed. The MAF used for each type of crops is shown in Table 11. These crops groupings are defined as in the Exposure Factors Handbook, with exposed fruits and vegetables defined as those that the edible portion grows aboveground without a protective rind or pod (*e.g.*, leafy greens, apples) and protected as those that the edible portion grows aboveground with a rind or pod that is not eaten (*e.g.*, peas, oranges). Root vegetables include tubers and roots, for which the edible portion grows underground (*e.g.*, carrots, potatoes).

Table 11. Moisture Adjustment Factors by Type of Produce

Model Code	Description	Exposed Fruit	Exposed Vegetables	Protected Fruit	Protected Vegetables	Root Vegetables	Reference
MAF	Moisture adjustment factor (% water)	85	90	87	81	81	EFH:2011 (US EPA, 2011)

Field Studies with Biosolids-Amended Soil

Yoo et al. 2011

Overview: This study collected grass samples from fields near Decatur, Alabama, that had received applications of sludge from a WWTP contaminated by industrial releases. There was no known irrigation at these sites. The study evaluated three grass species: Kentucky blue grass, Tall Fescue, and Bermuda grass. Each of these grasses could be forage for animals in pastures or used for hay or silage production. Soil and plant samples were collected at least several months after the last sludge application.

Results: The study presented soil to plant BCFs. The BCF values are labeled as grass soil accumulation factors (GSAF) for each of the grasses across multiple plots in terms of dry weight plant concentrations over dry weight soil concentration. The table below presents the results from 5 plots of grass, and a mean over all the grasses with tall fescue being weighted more heavily as it was in 3 of the 5 plots.

Table 12. Plant BCFs from Yoo et al. 2011

Plant Species	PFOA Plant BCF	PFOS Plant BCF
Kentucky blue grass	0.27	0.083
Tall Fescue (average of 3 plots)	0.29	0.076
Bermuda grass	0.13	0.035
Mean over all grasses	0.25	0.07

Uncertainties: Yoo et al. evaluated plants relevant to livestock consumption and evaluated the uptake factor in a farm field that was contaminated with PFAS due to biosolids application. It is possible that some degradation of PFOA or PFOS precursors occurred within the plant or soil that could lead to an over-estimation of plant BCF. The study reported that all FTOHs were nondetectable; this provides some indication that precursor conversion at this site may be a small effect. Given that uptake factors from biosolids-amended fields are considered the most relevant to risk assessment for the farm family, uncertainty related to the presence of precursors is unavoidable as many precursors cannot be quantified by available lab methods. The study also does not give a clear description of the distribution or total number of plant samples per species or soil samples from the fields taken in the study to create the uptake factors.

Blaine et al. 2013

Overview: This study evaluated tomatoes, lettuce, and corn in a midwestern US field that had been fertilized with biosolids at multiple rates. The study does not specify if fields were irrigated or the PFAS profile of irrigation water.

Results: The only field that produced measurable data for PFOA and PFOS was a pilot field that had biosolids applied at 4 times the agronomic rate. This field had PFOS soil concentrations of 13.9 ppb and PFOA soil concentrations of 5.2 ppb. All corn grain and tomato fruit samples had concentrations of PFOA and PFOS below the limit of quantification (LOQ), which were 0.2 ng/g for PFOA and 0.1 ng/g for PFOS. Within that field, the BCF for soil to lettuce (phrased as a BAF in the study) was 0.10 for PFOS. BCFs for PFOA were not quantifiable in any of the produce and PFOS data was only measurable in the lettuce. The PFOS value from this study for lettuce likely serves as a confirmation that the vegetative parts of plants will take up PFOS and that the BCF may be in the range from 0.07 to 0.10.

Uncertainties: The only fields with detectable amounts of PFOS in the plants received biosolids applications above the agronomic rate. The increased application rates were necessary to raise the contamination levels in the plants above analytical detection limits, but it isn't clear if the increased nutrients (N and P) in the soil would increase or decrease the plant uptake factor (BCF). As with any study that uses field applied biosolids, it is possible that there were precursors present that could breakdown to PFOA or PFOS and which increased the BCF.

The Yoo and Blaine studies both meet the qualifications for the highest tier of data for evaluating risks due to land application of biosolids and are particularly relevant for pasture scenarios. For pastured livestock (e.g., cattle, chickens, pigs, sheep) the uptake of PFOA and PFOS into grasses indicate that consumption of meat, dairy, and eggs could be a significant pathway of human exposure to these chemicals for farm families or CSA purchasers. The above studies provide data for parameterizing forage and silage for this pathway.

Greenhouse Studies Using Biosolids-Amended Soil

Uncertainties: For the following greenhouse studies, the BCFs calculated were typically higher than field studies and may be overestimated. Plants in pots for greenhouse studies have indicated higher uptake factors than field studies, this may be due to the roots having higher exposure to soil in the contained pot as opposed to a field. Although the BCFs may be elevated, it is still thought that the relative concentrations amongst the plant compartments in greenhouse studies indicate a pattern that would be representative of plants grown under field conditions. These studies also use biosolids-amended soil which may contain precursors.

Blaine et al. 2013

Overview: This greenhouse study (also cited above) investigated PFOA and PFOS fate in lettuce and tomatoes raised in biosolids-amended soil.

Results: The BCF values (1.34 to 2.52) for PFOA in lettuce grown under greenhouse conditions were much larger than the value cited for grasses (above). Greenhouse values for BCFs of PFOS in lettuce ranged from 0.32 to 1.67 and may indicate that greenhouse studies over-estimate field values for the same crop group, although for some shorter PFAS chain lengths in this Blaine study the field and greenhouse values are similar between the greenhouse and field studies.

Blaine et al. 2014

Overview: This study focused on biosolid-amended soil used in greenhouse studies of radishes, celery, tomatoes, and peas.

Results: The BCF values for soil to root concentration are significantly higher than the values for plants in the field studies mentioned above (Yoo et al., 2011 and Blaine et al., 2013). While this BCF value may be a valid indication of increased concentration in the roots of plants, it has not been confirmed by a field study. An important result of the greenhouse study, which is consistent with field studies, is that uptake into fruit is much lower than into roots or shoots, indicating that the presence of PFOA and PFOS in the edible fruit/seed portion of plants like tomatoes and peas may be of lower concern than consumption of edible greens (celery, lettuce, spinach, etc.) or roots (carrots, radishes). That the concentration is lower in edible portion of plants is perhaps unsurprising for long chain PFAS like PFOA and PFOS because the chemical must be transported across more membranes to enter the seeds, grains, and fruits of a plant.

Wen et al. 2016

Overview: This study focused on the role of protein content differences between tissues on the transport of PFAS within plants. This study illustrated the limitations of using plant uptake factors from greenhouse studies for risk assessment of farm crops.

Results: The uptake factors for PFOS and PFOA were several times larger than those calculated in the field studies above. For example, lettuce had a BCF for the shoot of 1.18 for PFOA and 0.396 for PFOS. However, the positive correlation between the uptake factors and the total protein content of the shoots and roots (higher uptake factors for tissues with higher protein content) is an interesting factor to consider for which plants may pose the most concern for human or livestock consumption. As measured in soil and fish, protein levels in distinct plant tissues may indicate where PFAS will preferentially partition within a specific crop group. The data on radish from this study are part of the range for determination of the root concentration factors for PFOA and PFOS.

Lechner and Knapp 2011

Overview: Lechner and Knapp 2011 employed greenhouse conditions to grow carrots, cucumbers, and potatoes in biosolid-amended soils, and the highest transfer factors for PFOA and PFOS were for the vegetative portions of each crop.

Results: The significantly larger plant uptake of PFOA and PFOS measured in greenhouse studies could be of concern for farms that utilize greenhouses for year-round marketing or for home gardeners that use a mixture of soil and biosolids in their greenhouse and potted plants.

Field Studies with Non-Biosolids Sources of Contamination

Since data were available to estimate needed parameters in the first two higher tiers of the data hierarchy, none of the studies from this tier were used in the risk assessment. Zhang et al. (2020) and Liu et al. (2019) focused on vegetables raised in fields that were directly impacted by their proximity to a PFAS manufacturing site. A summary of these data is presented by Li et al., 2022. The plant uptake values from these studies are frequently much larger, sometimes by over an order of magnitude, than the data available from the other literature sources cited above in this section. Given that there is a possibility that air deposition and irrigation water contamination stemming from the nearby PFAS manufacturing facility are increasing the concentrations of PFAS in the plants measured in these studies, these data are not appropriate for biosolids risk assessment. Specifically, Fig. 2.I of Li et al. (2022) presents a summary of the data from Zhang et al. (2020) and indicates that the BCF values for plants ranged from 0.5 to 31 for PFOA. A plant BCF value of 31 for PFOA in zucchini is amongst the highest reported in the literature. These high BCF values may be due to multiple exposure pathways in addition to soil for PFAS in a field adjacent to a PFAS manufacturing site, e.g., contaminated water or air deposition.

Scher et al. (2018) measured uptake of PFAS in garden produce at homes with contaminated irrigation water as the source of PFAS. Uptake factors from soil to plants for PFOS were consistent across leaves, fruits, and roots, ranging from 0.01 to 0.04. The uptake factors for PFOA were significantly higher with values for spinach as high as 1.4, but with most uptake factors ranging from 0.1 to 0.7.

BCF Selection for Plants

All selected values were based on studies using biosolids-amended soil. However, field data were available only for forage, silage, and above ground vegetables for PFOS. The remaining categories rely on greenhouse studies using biosolids-amended soils to grow plants in pots. For forage and silage, this assessment will use the mean BCF calculated across all the grasses in the Yoo et al. (2011) field study. This mean was used to represent the plant BCF for vegetative parts of plants that are common to forages and silages. For above ground vegetables (whether exposed or protected[12]), this assessment will use the single field value (for lettuce) available for PFOS from the study of Blaine et al. (2013), and a median of greenhouse values from Blaine et al. (2013, 2014) for PFOA, for which no field data were available. For fruits (whether exposed or protected), this assessment will use the median of detected greenhouse values from Blaine et al. (2013, 2014) and Lechner and Knapp (2011). Finally, for root vegetables, this assessment will use the median of detected greenhouse values from Blaine et al. (2014), Lechner and Knapp (2011), and Wen et al. (2016). The units for all the parameters below are dry weight crop concentrations divided by dry weight soil concentration which results in a unitless BCF plant uptake factor.[13] These values are summarized in Table 13.

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[12] "Protected" means that the edible part of the plant is covered (e.g., orange) while "exposed" means that typically the exterior of the fruit or vegetable is eaten (e.g., apple).

[13] Note that these values were converted to wet weight soil concentration for use in the model, and Appendix B presents them in those units, so the values differ somewhat. The conversion assumed a field capacity of 0.22 and a porosity of 0.43 (also used elsewhere in the models), resulting in a dry soil mass fraction of 0.87. The values based on wet weight soil in Appendix B were calculated by dividing the values above by that dry soil mass fraction.

Table 13. Selected Plant BCFs

Plant Type	Chemical	Plant uptake BCF (unitless)	Basis	Source
Forage	PFOA	0.25	field	Yoo et al. (2011) for grass
	PFOS	0.07	field	Yoo et al. (2011) for grass
Fruit	PFOA	0.11	pot	median or geomean of tomatoes from Blaine et al., 2013, sugar snap peas from Blaine et al. (2014), and cucumbers from Lechner and Knapp (2011)
	PFOS	0.03	pot	Sugar snap peas from Blaine et al. (2014) – only detected value for PFOS
Root Vegetables	PFOA	0.6	pot	median of pot carrots, potatoes, radish from Lechner and Knapp (2011), radish from Blaine (2014), and radish from Wen (2016)
	PFOS	0.7	pot	median of pot carrots, potatoes, radish from Lechner and Knapp (2011), radish from Blaine (2014), and radish from Wen (2016)
Silage	PFOA	0.25	field	Yoo et al. (2011) for grass
	PFOS	0.07	field	Yoo et al. (2011) for grass
Vegetables (above ground)	PFOA	1.3	pot	median of pot celery from Blaine et al (2014), pot lettuce industrial biosolids, and pot lettuce municipal biosolids from Blaine et al. (2013).
	PFOS	0.1	field	field lettuce from Blaine et al. (2013) – only field study for vegetables with a detected value

2.9.3.5 Livestock Uptake Factors

There are no existing EPA, FDA, or US Department of Agriculture (USDA) assessments that include livestock BTFs for PFOA and PFOS. In the context of agricultural risk assessment, BTFs are defined as the ratio of the concentration in the final product (*i.e.*, meat, milk, eggs) to the total intake rate of that chemical by the animal, represented in units of day per kg of food product. If a BTF is higher, this indicates that the animal transfers or accumulates larger amounts of the chemical into the tissue that becomes the finished food product. Because no existing source of BTFs was available for PFOA and PFOS, the EPA reviewed the available literature, assessed the available studies, calculated BTFs from the data provided in these published studies, and selected the most representative BTF for PFOA and PFOS in each food product. If more than one high-quality BTF was available for a food type, the median BTF was used. The following literature search strategy was used:

Database searched: PubMed

Search string: Title/Abstract search, (“perfluoroalkyl substance*” OR “polyfluoroalkyl substance*” OR “PFAS” OR “PFOA” OR “perfluorooctanoic acid” OR “perfluorooctanesulfonic acid” OR “PFOS” OR “perfluorooctane sulfonic acid”) **AND** (“livestock” OR “chicken*” OR “hen” OR “cattle” OR “cow” OR “cows” OR “swine” OR “pig” OR “pigs”) **AND** (“uptake” OR “accumulation” OR “transfer” OR “bioaccumulation” OR “biotransfer” OR “toxicokinetic*”)

Date searched: 8/1/2024. No date limitations on results.

Results: 58 studies

The following criteria must be met:

- Measured PFOA or PFOS concentrations in the exposure media
- Measured PFOA or PFOS concentrations in the finished animal product (*i.e.*, meat, milk, eggs)

- Durations of exposure relevant to common agricultural practices and environmental exposures (*i.e.*, durations that reach steady state or replicate the typical lifespan of the livestock before slaughter)
- Exposures in a media relevant to environmental exposures (*i.e.*, water, feed, soil)

The following criteria are preferred:

- Known rather than estimated intake rates of contaminated media
- Larger sample sizes

The results of this BTF selection effort are described below.

Eggs and Chicken Meat

Wilson et al. 2020

Overview: This study was a controlled laboratory study that included 119 laying hens. All hens were 30-weeks old at the beginning of the study period. The hens were divided into 5 groups (22 hens in the control group and lowest concentration treatment group and 25 hens in remaining 3 treatment groups). Hens were exposed to PFOA, perfluorohexane sulfonic acid (PFHxS), PFOS, and PFHxA via drinking water at concentrations of 0, 0.3, 3, 30, and 300 µg/L for 61 days. Eggs were collected throughout the treatment period. At the end of the treatment period, the treatment group hens were given PFAS-free water for 30 days. Eggs were also collected and analyzed during this depuration period. No meat samples were collected in this study.

Results: No negative health, welfare, or behavioral changes in the hens over the course of the study were noted. A subset of eggs was sampled to analyze the relative distribution of PFAS in egg yolk and albumen (egg white). Over 99% of the PFOA and PFOS present in eggs were distributed to the egg yolk rather than the albumen, consistent with data of Kowalczyk et al., 2020. For all hens in the treatment groups, egg concentrations of PFOA and PFOS increase until days 24-30. After this initial increase, PFOA and PFOS concentrations in eggs reached apparent steady state until the cessation of treatment.

PFOA concentrations in whole eggs ranged from 500 to 400,000 ng/kg and for PFOS ranged from 800 to 1,000,000 ng/kg, with concentrations proportional to dose. Daily intake rates ranged from 40 to 47,000 ng/day for PFOS and 57 to 54,000 ng/day for PFOA, depending on the treatment group. Biotransfer factors (BTFs) were calculated for each treatment group using the calculated average intake rate and the average egg concentration during the steady-state period. The average BTF for PFOS was 21 day/kg (ranges from 19-24 day/kg) and the average BTF for PFOA was 8.6 day/kg (ranges from 8.1-9.2 day/kg).

Uncertainties: This study was well controlled with limited uncertainties. Though there were several quality control metrics included in the study, this study still includes some uncertainty in the PFAS exposure for the treated hens. Hens were only included in this study if they had non-detectable levels of the four studied PFAS in their eggs prior to the start of the study, ensuring that no prior exposure was impacting the results. The feed and bedding material for the animals was tested and confirmed to be free of the studied PFAS. Because this study used drinking water for exposure that was prepared in-lab, it is known to not contain precursors to PFOA and PFOS. Overall, this is a high-quality study with a large sample size.

Kowalczyk et al. 2020

Overview: This was a laboratory study which included 12 laying hens. The hens were 6 months old at the beginning of the study and were fed a combination of highly contaminated hay (harvested from a field that received contaminated biosolids and paper-derived compost in southern Germany) and barley for 25 days. The barley and hay were analyzed for 14 PFAS, and TOP assay (Göckener et al., 2020). TOP assay converts oxidizable precursors of PFOA and PFOS to PFOA. Of the 14 PFAS analyzed, 12 were below the limit of quantification in the hay and barley. The PFOA concentrations were 0.8 µg/kg in barley and 287 µg/kg in hay, for a combined average intake rate of 0.6 µg/day per hen. The PFOS concentrations were 2.5 µg/kg in barley and 1,654 µg/kg in hay, for a combined average intake rate of 2.8 µg/day per hen. After TOP analysis, PFOA levels of the mixed feed increased 786%; again, note that TOP analysis oxidizes PFOA and PFOS precursors to PFOA. After the 25-day feeding period, 4 hens were slaughtered and 8 were fed a non-contaminated diet until study day 67. At this point, the remaining hens were slaughtered. Muscle, liver, and kidney samples and egg yolks were then analyzed.

Results: This study did not record any treatment related adverse health effects in the hens. In the subset of eggs where both yolk and albumen were tested, over 99% of PFOA and PFOS present in egg were measured in the egg yolk. Over the duration of the exposure period, concentrations of PFOA and PFOS increased rapidly from days 0-10, with slower increases for the remaining 15 days of the exposure period.

At exposure day 25, the average PFOA concentration in egg yolks was 18.6 µg/kg wet weight (ww), which corresponded to an average total egg concentration of 5.2 µg/kg ww. At exposure day 25, the average PFOS concentration in egg yolks was 560 µg/kg ww, which corresponded to an average total egg concentration of 157 µg/kg ww. Using the reported feed intake rates, egg BTFs were calculated for PFOA at 8.7 day/kg and for PFOS at 56 day/kg. After TOP assay of the egg yolks, PFOA concentrations increased 647%.

At exposure day 25, the average PFOS concentration in hen muscle was 6.2 µg/kg ww and the average PFOA concentration was 0.3 µg/kg ww. These concentrations correspond to a muscle BTF of 2.2 day/kg for PFOS and 0.5 day/kg for PFOA. Hen liver concentrations for PFOA and PFOS were significantly higher than muscle concentrations (3.7 µg/kg ww for PFOA and 72.3 µg/kg ww for PFOS). TOP assay was not performed on muscle samples.

Uncertainties: This study has several limitations that could influence how the results are interpretated. The hens in this study were exposed for 25 days. In Wilson et al. 2020, the daily egg concentrations of PFOA and PFOS increased during the beginning of the treatment window and did not stabilize until treatment day 24-30, depending on the treatment group. In the Kowalczyk et al. (2020) study, egg concentrations appeared to level-off after day 10 of exposure but continued to trend fractionally upward until the end of the treatment phase on day 25. For this reason, the relatively shorter exposure duration in this study could result in a slight underestimate of the BTF. The exposure media in this study also contains significant concentrations of precursors, a fraction of which appeared to transform to PFOA or PFOS in the hens or eggs (as previously described, the TOP analysis increases PFOA concentrations 786% in feed, but only 647% in eggs, which indicates that though the majority of precursors are transferred to eggs intact, a fraction appear to have transformed to their terminal degradates of PFOA or PFOS). The presence of precursors in feed could thus result in a slight overestimate of BTFs. Although this study has some uncertainties, the overall study quality is sufficient for quantifying BTFs in eggs and meat.

BTF Selection for Eggs and Chicken

Eggs: Either the Wilson et al. (2020) study or the Kowalczyk et al. (2020) study would be sufficient for estimating BTFs in eggs. The fact that PFOA BTFs from both studies are nearly identical (8.6 and 8.7) increases confidence in these results. The PFOS BTF calculated from data of Kowalczyk et al. (2020) were approximately three times higher than those calculated from data of Wilson et al. (2020) (21 compared to 56). Kowalczyk et al. (2020) hypothesized that this discrepancy is due to the presence of significant levels of PFOS precursors in the hay used in the study. It is possible that some PFOS precursors present in the feed used in Kowalczyk et al. (2020) readily biotransformed to PFOS in the hens, while other PFOA or PFOS precursors are passed to the egg yolk without transformation. Given that the Wilson et al.

(2020) study has a significantly larger sample size than Kowalczyk et al. (2020) and that this study does not have the compounding variable of precursors in the feed ingredients, this assessment will use the BTFs from Wilson et al. (2020) of 21 day/kg for PFOS and 8.6 day/kg for PFOA.

Meat: Kowalczyk et al. (2020) is the only study available to quantify BTFs for PFOA and PFOS in chicken meat. There was significantly less transfer of PFOA and PFOS to laying hen muscle compared to egg yolk, which aligns with other studies reporting lower PFOA and PFOS concentrations in chicken meat than in eggs (Braunig et al., 2017; EFSA CONTAM Panel, 2020; Lasters et al., 2023). Importantly, this study only analyzes meat from laying hens, which is feasibly consumed by those with backyard flocks, but generally used commercially only in making processed foods and canned soup. Broiler chickens, which are grown for meat production and are generally slaughtered at 6-10 weeks after hatching, may have different accumulation rates than laying hens. For example, the elimination pathway of egg laying is not available to broiler chickens and all laying hens are female, while broiler chicken flocks contain both sexes. Chickens raised for meat also have a shorter lifespan than laying hens. The USDA is currently conducting a PFAS uptake study on broiler chickens; until these data are available, the chicken meat BTFs presented in this assessment have uncertainties when applied to meat consumed from animals other than laying hens. Though Kowalczyk et al. (2020) may overestimate the PFOS BTF due to known contamination of feed with PFOS precursors, the study is nevertheless of sufficient quality to quantify a BTF for this assessment. The BTF is 0.2 day/kg for PFOA and is 2.2 day/kg for PFOS. Again, the current BTFs, calculated from laying hens, are appropriate for this assessment scenario where a farm family primarily has hens for egg production, but occasionally slaughters hens for food (for example, at the end of their laying life). This scenario is not relevant to commercial food operations raising broiler hens for meat production. This assessment is also not considering intake and exposure from consumption of animal livers; given the elevated levels observed in livers, this may be an important pathway of exposure for those who consume liver. More data are needed on PFOA and PFOS uptake into breeds of chickens more typically used for meat.

Beef and Milk

Vestergren et al. 2013

Overview: Vestergren et al. (2013) reported an observational study of milk and meat from a dairy farm in Sweden. The farm had 92 Swedish Red dairy cows of varied ages that had consumed silage, corn, and barley grown on the farm and drinking water from a groundwater well. The farm was not known to have any PFAS point sources (such as contaminated sludge application) and was selected as a representative “background” dairy farm for Sweden, meaning that PFOA and PFOS contamination is thought to be caused only by long-range atmospheric transport and deposition. The cows at this dairy were mainly confined to a barn but were allowed to graze on a pasture during the summer months. This study did not quantify PFAS soil concentrations and milk was only sampled during the months that the animals were confined to the barn. The average intake of PFOA and PFOS for the cows was estimated using feed PFAS concentrations and consumption rates of silage, corn, barley, and water. Pooled milk samples were collected from the milk storage tank monthly (from November to April), on the same days when representative samples of feed were collected. Over the course of the study, five cows were slaughtered, and muscle, liver and whole blood were analyzed for PFAS. Given that the animals had been living on the contaminated farm for their entire lives, PFAS concentrations in the animals are assumed to be at steady state.

Results: Daily intake rates were estimated to be 613 ng/day for PFOA and 303.6 ng/day for PFOS, based on the measured PFOA and PFOS concentrations in feed, water, and supplements and assumed consumption rates for each category. The supplements at this dairy were purchased from a supplier and found to have no detectible PFOA and PFOS, but the authors assumed that supplements contained PFOA and PFOS at ½ the MDL for each analyte. PFOA and PFOS concentrations in pooled milk samples were 6.7 and 6.2 ng/L, respectively. Muscle of the five slaughtered cows contained, on average, 7 ng/kg for PFOA and 21 ng/kg for PFOS. The milk BTFs were 0.01 day/kg for PFOA and 0.02 day/kg for PFOS. The muscle BTFs were 0.01 day/kg for PFOA and 0.07 day/kg for PFOS.

Uncertainties: This study site is a farm where there are no known proximal sources of PFAS contamination (and no known history of sludge application to the pastureland), indicating that the source of PFAS is long-range atmospheric transportation and deposition; the study included measurements of PFOA, perfluoroheptanoic acid (PFHpA), perfluorohexanoic acid (PFHxA), PFNA, PFDA, PFUnDA, PFDoDA, PFHxS, and PFOS. It is likely that PFOA and PFOS precursors were present in the water, feed, and soil around this farm, but it is unknown the degree to which precursors could impact the calculated BTFs. If precursors to PFOA and PFOS were present in the feed, water, and/or soil, this would result in overestimation of the BTFs. This study also did not attempt to quantify the average annual intake from soil that cows consume during the summer months when cows were pastured. Not including soil as a potential intake pathway could result in overestimated BTFs. The intake rates in this study are also somewhat uncertain because they are estimated from the farmer's assumptions about the intake rates for his cows (for example, the farmer communicated that his cows consume about 50 L/day of drinking water). Though this study is large for milk (92 cows' milk was pooled and analyzed), only 5 cows were slaughtered for the muscle analysis. Overall, this study is of sufficient quality to use quantitatively in the assessment but includes some areas of uncertainty that would likely result in overestimates of BTFs for dairy cows.

Kowalczyk et al. 2013

Overview: Six lactating Holstein cows housed at the German Federation for Risk Assessment were fed a PFAS-contaminated diet for 28 days. After the exposure period, three cows were slaughtered and the remaining three were fed a PFAS-free diet for an additional 21 days. The diet was mixed from PFAS-contaminated grass silage and hay harvested from a contaminated farm in Lower Saxony (the same materials used in Kowalczyk et al., 2020). The cows were housed in individual tie-stalls and their intake of feed was quantified each day. Meat and milk samples were analyzed for PFOA, PFOS, PFHxS, and perfluorobutanesulfonic acid (PFBS). The serum half-life of PFOS in beef cattle was estimated to be 116 days (Lupton et al., 2015); with an exposure of only 28 days, PFOS concentrations in muscle and milk in this study are not expected to represent steady-state concentrations. In contrast, the estimated half-life of PFOA in cattle is estimated to be 19 hours (Lupton et al., 2012) and dairy cows having lifetime exposures to PFAS contaminated feed and water appeared to have rapidly cleared PFOA, as evidenced by no accumulation above quantification limits in serum (Lupton et al., 2022).

Results: There were no noted adverse effects in the cows over the duration of the study. Average PFOS concentrations measured in grass silage and hay were 200 and 1,924 µg/kg while average PFOA concentrations in grass silage and hay were 79.3 and 333 µg/kg. Consumption of grass silage and hay were 8.9 and 1.4 kg/day, respectively corresponding to an average intake rate of 1,172 µg PFOA/day and 4,472 µg PFOS/day. Note that the PFOA and PFOS concentrations in Table 1 of Kowalczyk et al. (2013) are switched; this mistake has been confirmed with the study's lead author. During the exposure period, PFOS concentrations in milk increased at a steady rate. Once the exposure period ended, the three cows fed a PFAS-free ration had milk PFOS concentrations similar to the level reached on the last day of exposure, indicating that depuration of PFOS in milk was slow. During the exposure period, PFOA concentrations in milk did not exceed the LOD (0.1 µg/L) until around exposure day 10. PFOA concentrations in milk then hovered near the LOD until the exposure period ended. In cows fed PFAS-free feed after exposure, PFOA concentrations in milk were non-detectable. For PFOS, mean milk concentrations were calculated using the milk from the last day of the exposure period and the depuration period (study days 29-45). Mean PFOS concentrations in milk were 32.1 µg/L. This resulted in a BTF of 0.007 day/kg. PFOA was only detected in milk during the latter part of the exposure period, and not in milk collected during the depuration period. Using the average detected concentration of PFOA in milk during the exposure period, a BTF of 0.00006 (6×10^{-5}) day/kg is calculated. Thus, this study indicates that PFOA had very limited accumulation into milk over the given exposure period.

The average PFOS muscle concentration in the three animals slaughtered after the exposure period was 145 µg/kg for PFOS. After the exposure period ended, the PFOS concentration measured in muscle tissues increased to 178 µg/kg. The average muscle concentrations of all animals (slaughtered at study day 29 and 45) was 161.5 µg/kg. The BTF for PFOS in muscle calculated using this average value is 0.036 day/kg. The average PFOA muscle concentration measured in the three animals slaughtered after the exposure period was 0.6 µg/kg. PFOA was not detectable in the remaining three animals slaughtered after the depuration period. A BTF calculated using only the three animals slaughtered on study day 29 is 0.00006 (6×10^{-5}) day/kg. Thus, this study also indicated that PFOA had very limited accumulation in dairy cow meat.

Uncertainties: The 28-day PFAS exposure period in this study is not long enough for PFOS to reach steady-state concentrations; Lupton et al. (2012) demonstrated that steady state concentrations of perfluorosulfonic acids (PFSAs) in dairy animals were not met until after about 1.5 years of exposure. Therefore, BTFs calculated from this study will underestimate accumulation of PFOS. PFOA concentrations in milk and meat in this study were below, or near, the limits of detection, which indicates that PFOA does not significantly accumulate in meat or milk of dairy cows and adds to the uncertainties of these values. The feed used in this study is the same feed that is used in the Kowalczyk et al. 2020 chicken study; it is known that this feed contained significant concentrations of PFOS precursors. The presence of PFOS precursors in the feed may explain why PFOS concentrations in milk and meat are elevated even after the exposure period ends. Another potential explanation of this observation is that PFOS can be stored in other compartments of cattle, like skin, which could result in ongoing excretion through milk even after exposure through feed and water has ended (Lupton et al., 2022). The presence of PFOS precursors in feed would result in underestimations of BTFs. Finally, this study has a relatively small sample size of six animals.

Drew et al., 2021; 2022

Overview: Drew et al. (2021) reported on the accumulation of PFAS in Belted Galloway beef cattle and mixed breeds of sheep raised for meat on a hobby farm in Australia that had water contaminated with AFFF from a nearby facility. This discussion will focus on accumulation results for cattle. The study was split into two phases, each approximately one year long, with one year between phases. The first phase occurred before remediation activities were taken to reduce PFAS levels in the livestock drinking water by attempting to divert contaminated water from the neighboring property away from the farm. The second phase took place after this remediation activity. The only source of feed for the cattle during the duration of the study was forage.

During phase one, soil (n = 36) and grass (n = 5) samples were collected from the forage area. Drinking water was measured two times, at the beginning of phase one and near the end of phase one; this sampling only monitored for four PFAS and is not reported. Serum levels from 5 cows (9-14 years old) and 9 cattle (2-22 months old) were collected.

During phase two, the stock water for the cattle was sampled again, and serum from 19 cattle were collected (all adults over 1.5 years old, three were steers and the remaining were heifers and cows). The 19 cattle included in phase two were moved to a research facility 19 days before the last blood sampling event in this study. At the research facility, five of the animals had PFAS blood monitoring for 214 days post removal from the farm; 11 animals were euthanized on day 63 post removal from the farm and

PFAS levels were measured in tissues (these results are presented in a different study, Drew et al., 2022). Using the tissue concentration data and the elimination half-lives, PFAS levels in tissue could be estimated for the animals at the time of transfer from the farm. The water remediation activity was completed Fall of 2016, and the serum samples were collected in Spring of 2018; the authors assume that PFAS concentrations in the animals are at steady state throughout phase two of blood sampling.

Results: There were no adverse effects in cattle or sheep reported in this study. Phase one soil sampling found that PFOS had a mean concentration 0.003 mg/kg dry weight (3 µg/kg dry weight) in soil; PFOS was not detected at quantifiable concentrations in grass (LOQ = 0.0005 mg/kg wet weight, equivalent to 0.5 µg/kg). PFOA was not present at a measurable level in soil or grass. Water results from phase one are not presented in the publication. Because the accumulation calculations were only conducted on animals included in phase two, the discussion of results will focus on water and serum levels for those 19 cattle.

During phase two, the mean water concentrations were 3.0 µg/l (3,000 ng/L) for PFOS and 0.87 µg/L (870 ng/L) for PFOA. Water concentrations of PFOA and PFOS did not significantly differ between the two sampling dates. During the phase two study (while the cattle are on the contaminated farm), serum levels of PFOS range from 275 to 455 ng/mL while PFOA was consistently non-detectable in all serum samples. Drew et al., 2019 (the companion study reporting data collected at the research facility) found that the serum half-life for PFOS in these cattle was 74 days. This study also found that the partitioning coefficient from serum to muscle for PFOS was 0.072 ± 0.02 on day 62 (the transfer day to the research facility) and 0.08 ± 0.03 on day 215. The authors of this study calculate steady-state serum concentrations of 436.2 ± 59.0 ng/ml. Using the steady-state serum concentration, the median muscle partitioning coefficient (0.076), the mean concentration in drinking water, and an assumed drinking water consumption rate of 59.8 L/day (Drew et al., 2021), a biotransfer factor of 0.18 day/kg is calculated for PFOS. This BTF assumes that all intake is derived from contaminated drinking water. No BTF is derived for PFOA because the study finds that there is no measurable accumulation of PFOA into cattle serum.

Uncertainties: Overall, there are some uncertainties in this study from the lack of precise information on the amount of PFOS intake in the 19 phase two cattle used to derive BTFs. Because the pre-remediation activity drinking water concentrations are not presented, it is not clear if residual PFOS loading from original drinking water source could be continuing to impact the serum levels of phase two cattle. The transfer factor calculation also assumes negligible intake of PFOS from grass and soil; this assumption may lead to an overestimate of BTF. The BTF calculation also assumes a drinking water intake for the cattle based on the climate and weight of the cows, rather than a measured drinking water intake. Overall, this study is of sufficient quality to calculate BTFs.

Johnston et al. 2023 & Lupton et al., 2022

Overview: Johnston et al. (2023) and Lupton et al. (2022) measured blood, ear notch (skin), and muscle PFAS concentrations in dairy cattle from a farm containing AFFF-contaminated drinking water in New Mexico. Silage at this farm was also sampled and confirmed to contain PFAS. Blood and ear notch samples were collected from 175 cattle on the farm. Thirty of these cows (10 heifers, 15 lactating, and 5 dry) were moved to an uncontaminated research facility (New Mexico State University). Two weeks after the move, 20 of the cows were euthanized and necropsied, with blood plasma and tissues analyzed. In the remaining 10 cows, blood samples were collected every two weeks. The two oldest cows died during the study period. Finally, the 8 remaining cows were euthanized and necropsied at either 137 or 153 days after arrival to the research facility. In all, paired blood and muscle data are available from 28 cows.

Results: Though two cows died during the course of the study, these were the oldest cows in the cohort and there were no reported adverse effects in the cows. At the contaminated farm, the mean concentration of PFOS was 818 ng/kg in water and in 3,482 ng/kg in silage. PFOA results are not reported for the contaminated media at the farm or in the blood and muscle samples. This study does not report an observed or estimated feed or water consumption rates for the cows on this farm. To calculate the total intake of PFOS through silage and water, the reported silage consumption rate in Vestergren et al. (2013) is used (38.5 kg/day) and the general estimated drinking water intake for dairy cows is used (92 L/day, US EPA 2003g). This calculation estimates a total PFOS intake from silage and water to be 209,313 ng/day.

Serum levels of PFOS in the cows decline over time as the animals depurate PFOS at the research facility. There is a log-linear relationship between PFOS levels in plasma and muscle. The total PFOS concentrations in plasma and muscle in each of the 28 cows with this data available are reported in Supplemental Information Table B (in this table, the heifers are reported as "young"). These data are used to calculate partitioning coefficients between plasma and muscle for each cow. The partition coefficient from plasma to muscle ranges from 0.03 to 0.11. Dry and lactating cows have similar partitioning coefficients, but the heifers have lower PFOS partitioning to muscle (a smaller partitioning coefficient). Because most of the young cows have not reached a steady-state serum level, the average of partitioning coefficient excluding the young is used for further calculations. This mean partitioning coefficient is used to estimate the muscle concentrations on the dry and lactating cows using the plasma concentrations at the time of removal from the farm. The mean muscle concentration in these dry and lactating cows is 7.3 ng/g. Using this mean muscle concentration and the estimated PFOS intake rate, the BTF for PFOS in muscle in dairy cows is 0.035 day/kg.

Uncertainties: These studies did not include any PFOA results above a limit of quantification. This study is also focused on muscle uptake in dairy cows, which are not commonly used for beef production. As described previously, it is expected that there are significant differences in BTFs for dairy cows and cattle primarily used for beef consumption. Also, this study does not include information about the amount of water and feed consumed by the animals at this farm, and the estimate of total intake does not include any exposure from soil.

Chou et al. 2023

Overview: This study describes the development of a generic physiologically based pharmacokinetic (PBPK) model for adult beef cattle and lactating dairy cows useful for estimating tissue and milk distribution and depuration rates of PFAS. The generic beef cattle model consisted of four tissue compartments including liver, kidney, muscle, and the rest of the body (notably, the model does not include a compartment for plasma). The generic model structure for dairy cows is the same as the beef cattle model, but also includes udder and milk compartments. Physiological parameters, including body weight, cardiac output, fractions of blood flow to tissues, and the volume fractions of individual organs were collected from a previous review article that summarized published experimental data in various breeds of beef and dairy cattle (Swedish Red, Holstein Friesian, Belted Galloway, Australian Lowline, American Angus, and Japanese Black). Chemical-specific parameters included protein binding, absorption/elimination rate constants, partition coefficients, enterohepatic circulation, and renal reabsorption parameters. There were not chemical specific data for PFOA and PFOS in cattle and cows; instead, these chemical-specific values were parameterized using a previously published PBPK model for PFOA and PFOS in rats. The model can consider intake from soil and water. The model outputs muscle and milk concentrations over time. The final PBPK model was coded as a R-Shiny application and is available online.

Results: The authors validated their models against an independent PFOA and PFOS dataset in Chinese beef and milk that was not used to parameterize the model. Because these datasets included final food products and did not include information on exposure of the animals, the exposure was estimated using data describing water and soil PFAS concentrations in China. The model was also validated using the tissue results reported by Kowalczyk et al. (2013). The model was generally within a two-fold error range of the observed PFOA and PFOS concentrations in all compartments except for PFOA in milk, which was underestimated by the model (compared to PFOA concentrations observed in milk and yogurt products in China), and PFOS in muscle, which was overestimated by the model.

This model could be used to calculate BTFs by setting a concentration of PFOA and PFOS in soil and water, calculating intake rates using the consumption rates employed in the model, and comparing the estimated milk and muscle concentrations after two years of exposure to the calculated intake rates. Two years of exposure was selected as the time window to estimate BTFs because this is the age that beef cattle are slaughtered and that dairy cows generally enter milk production. This exposure time is also more than sufficient to reach steady state in cows and cattle (Lupton et al., 2012; Lupton et al., 2015). To simplify this calculation, the authors assumed that water was the only source of PFOA and PFOS exposure and set the water concentration to 2 µg/L for both chemicals.

Using these assumptions, the model estimated that PFOS in muscle after a 2-year exposure would be 12.2 ng/g in beef cattle and 65.9 ng/g in dairy cows. The modeled estimated PFOA muscle after a 2-year exposure would be 0.0253 ng/g in beef cattle and 0.166 ng/g in dairy cows. This results in muscle BTFs for PFOS of 0.09 and 0.30 day/kg in beef cattle and dairy cows, respectively. Muscle BTFs for PFOA are 0.0002 and 0.001 day/kg in beef cattle and dairy cows, respectively.

The model also calculated PFOA and PFOS concentrations in milk. After 2 years of exposure, the predicted PFOA concentration in milk was 0.03 ng/mL and the predicted PFOS concentration in milk was 24.4 ng/mL, resulting in milk BTFs of 0.0001 and 0.11 day/L for PFOA and PFOS, respectively.

Uncertainties: The BTFs generated from this publication's data are based on modeled, not measured concentrations in exposure media and animal products. The BTFs presented from this study therefore represent estimates from a PBPK model with uncertainties in many of the parameters. There are significant uncertainties in the results.

Xiao et al. 2024

Overview: This study measured PFAS concentrations in feed and raw milk from 92 dairy farms across 20 provinces of China. At 70 of these farms, the researchers were also able to measure PFAS in the cow's water. Researchers calculate the "carry over rate" (COR) for PFOA and PFOS, which is defined as the mass of the chemical eliminated through milk secretion divided by the mass of the chemical consumed through feed and water. This COR can be converted to a BTF by dividing the value by the assumed milk secretion rate (kg/day). This study calculated CORs using an assumed daily consumption rate of silage of 20.6 kg/d dry weight, an assumed drinking water consumption rate of 83.6 L/d, and an assumed milk production rate of 26.5 kg/d. For the 22 farms where no drinking water data were available, the researchers used the mean drinking water values for PFOA and PFOS in the farm's region to calculate a COR.

Results: PFOA water concentrations ranged from non-detect to 113 ng/L (mean of 5 ng/L, detection rate of 79%) and PFOS water concentrations ranged from non-detect to 18 ng/L (mean of 0.7 ng/L, detection rate of 53%). Feed concentrations ranged from non-detect to 10.6 ng/g for PFOA (mean of 0.7 ng/g, detection rate of 35%) and non-detect to 0.45 ng/g for PFOS (mean of 0.08 ng/g, detection rate of 47%). Raw milk concentrations ranged from non-detect to 500 ng/L for PFOA (mean of 80 ng/L, detection rate of 57%) and from non-detect to 160 ng/L for PFOS (mean of 20 ng/L, detection rate of 62%). The researchers did not provide COR results for each farm, instead reporting mean intakes, mean excretions, and mean CORs for each chemical. The mean COR for PFOA is 15.78, which equates to a BTF of 0.006 day/kg. The mean COR for PFOS is 29.58%, which equates to a BTF of 0.01 day/kg.

Uncertainties: This study has uncertainty in the intake rates because there are assumed consumption rates for food and water that are not specific to the farm or region and because 22 of the 92 included farms had assumed rather than measured drinking water concentrations. Because farm-specific data were not included in the publication, it was not possible to recalculate CORs for only farms with measured drinking water intakes. This study finds that the majority of intake to cows at the farms included in this study derived from feed rather than water, which reduces the impact of the uncertainties regarding drinking water exposures to the cows. This study does not include measurements of PFOA or PFOS precursors in feed, water, or milk. Overall, this study includes a large number of farms and finds similar BTFs for PFOA and PFOS as are derived from Vestergren et al. (2013).

BTF Selection for Milk and Beef

Milk: Kowalczyk et al. (2013) is not suitable as a basis for BTFs for PFOS due to the short exposure time of the study, which will result in an underestimate of PFOS accumulation in a farm scenario. This study also has PFOA levels in milk that are below or around the detection limit, leading to uncertainty. The Chou et al. (2023) PBPK model is parameterized using PFOA and PFOS-specific constants derived from rat studies. There are obvious physiologic and significant differences in these values between rats and cows. For example, the plasma half-life of PFOA in cattle is <24 hours (Lupton et al., 2012) whereas the plasma half-life of PFOA in male rats is 16 days (DeSilva et al., 2009). This modeling study is thus too uncertain to be used in deriving BTFs for the assessment. Both Vestergren et al. (2013) and Xiao et al. (2024) are potential candidates for BTFs in milk. Though the Vestergren et al. (2013) study has some uncertainties regarding the presence of precursors and potential impacts of soil ingestion from grazing, overall, this is the best available study for deriving BTFs because Xiao et al. (2024) includes assumed rather than measured drinking water concentrations for some of the farms that are included in the reported summary statistics. Notably, Kowalczyk et al. (2013) and Chou et al. (2023) indicate that PFOA accumulation in milk is close to zero, while Vestergren et al. (2013) and Xiao et al. (2024) find that PFOA accumulation in milk is only two-fold less than PFOS accumulation. Additional studies of PFOA accumulation into milk would improve our understanding of potential exposure risks for this pathway.

Beef Cattle: Much of the same rationale for study selection of BTFs for beef applies as did for milk. Given the limitations of Kowalczyk et al. (2013), Johnston et al. (2023) and Lupton et al. (2022), and Chou et al. (2023), the best studies for quantifying BTFs in beef are Vestergren et al. (2013) and Drew et al. (2021). Note that the Vestergren et al. (2013) study measured muscle concentrations from lactating cows, not cattle raised for beef production. As illustrated by Chou et al. (2023) and Drew et al. (2021), different BTFs would be expected for lactating cows and beef cattle, in part due to the added excretion pathway of milk production in lactating cows; In fact, Drew et al. (2021) finds significantly higher BTFs for PFOS in cattle than Vestergren et al. (2013) and Johnston et al. (2023) found for meat in dairy cows. Because Drew et al. (2021) is a high-quality study measuring uptake into breeds used for beef production, it is selected for the PFOS BTF in beef. Drew et al. (2021) did not find that PFOA accumulates to measurable levels in serum of cattle used for beef production. However, Vestergren et al. (2013) does find measurable levels of PFOA in beef from culled dairy cows. While extrapolating the PFOA BTF measured in dairy cows to more commercially relevant beef production settings introduces significant uncertainty, the PFOA BTF calculated from Vestegren et al. (2013) represents the best available estimate for PFOA uptake at this time. More studies are needed on the uptake of PFOA and PFOS into breeds typically used for beef production.

Pigs

Numata et al. 2014

Overview: Three groups of fattening pigs (10 gilts, 10 barrows, and 10 young boars) were housed at the German Federation for Risk Assessment. In each group 8 were fed a PFAS-contaminated diet and 2 were fed PFAS-free feed. Feed intake was restricted to 2 kg/day per hog with an exposure period of 21 days. The diet was mixed with PFAS-contaminated hay harvested from a contaminated farm in Lower Saxony (the same hay used in Kowalczyk et al. 2020). Representative feed samples were analyzed for PFAS content on five separate exposure days. Plasma samples were taken on five days throughout the exposure period and the day of slaughter (day 22). Urine samples were also collected sporadically throughout the sampling period, with an average of 2.5 urine samples collected per pig. Muscle, plasma, urine, and organs were analyzed for 12 PFAS. The results were used to parameterize a PBPK model.

Results: No adverse health impacts of test animals were reported by the study authors. Serum levels of PFOA and PFOS increased throughout the duration of the 21-day study. Based on the plasma measurements taken during the study, the authors estimate that the elimination half-life of PFOS is 634 days, significantly longer than the exposure timeframe of this study. The elimination half-life for PFOA is 236 days, also significantly longer than the exposure timeframe of this study. Given that the concentrations of PFOA and PFOS in serum did not level off during the exposure duration of the study, it is not possible to extrapolate tissue concentrations at 180 days, which is generally the time when pigs are slaughtered.

Uncertainties: BTFs calculated from tissue concentrations at day 22 of exposure would significantly underestimate uptake. Similarly, BTFs calculated using the PBPK model presented in this study would represent steady-state conditions, which were not reached by the time of slaughter. BTFs calculated using the PBPK would therefore significantly overestimate risk. Additional studies are needed to understand BTFs exposure durations expected in the conceptional model of an agricultural setting.

BTF Selection for Pork

Only one study was available in pigs and this study was not sufficient to calculate BTFs for PFOA and PFOS. For this reason, pigs are not included in the farming models in this assessment.

Overview of Livestock Uptake Parameters

Table 14. Selected Livestock BTFs

Livestock type	Product	PFOA BTF (day/kg)	PFOS BTF (day/kg)	Study
Chicken	Meat	0.2	2.2	Kowalczyk et al. 2020
Chicken	Eggs	8.6	21	Wilson et al. 2020
Cows	Beef	0.01	0.18	Vestergren et al. 2013 for PFOA; Drew et al., 2021 for PFOS
Cows	Milk	0.01	0.02	Vestergren et al. 2013

2.9.3.6 Livestock Dietary Intakes

The produce, meat, and milk exposures will be evaluated using the methodology found in HHRAP (US EPA, 2005), developed for hazardous waste combustion facilities. That methodology includes recommended input values for many, but not all of the livestock diets included in this assessment. For example, HHRAP does not evaluate water consumption by livestock, which was considered an insignificant pathway for combustor emissions. It is known that PFOA and PFOS can be present in groundwater and surface water, so that pathway was included in this analysis.

There are no data available on PFOA and PFOS bioavailability to livestock specifically from feed, water, or soil; this assessment assumes 100% is available when orally ingested. The studies used to derive BTFs for livestock include a variety of exposure scenarios for the experimental animals. In some cases, the animals are exposed through water only, in other cases the animals are exposed through feed only, and in other cases the animals are sampled from a pasture farm where they have exposure from feed, water, and soil. When comparing the PFOA BTFs derived for chicken eggs from the Wilson et al. (2020) study (animals exposed only through water) and the Kowalzak et al. (2020) (animals exposed only through contaminated feed), the calculated BTFs are nearly identical. This indicates that if there is a reduced bioavailability of PFOA in chicken feed, that effect is likely negligible. In the case of dairy cows, the BTFs selected for this study (from Vestegren et al., 2013) were derived by calculating the exposure from feed and water combined. If there were a reduced bioavailability of PFOA or PFOS in feed in dairy cows, this would already be factored into the BTF calculation. For beef cattle, the BTFs were also derived using data from pasture-fed cows (Vestegren et al., 2013 and Drew et al., 2021), so these factors also inherently consider differences in bioavailability between feed and water in the calculated values. Note that part of the reason previous assessments included assumptions about reduced bioavailability in feed compared to water is because the BTFs in these assessments were modeled, not measured. By using BTFs derived from empirical experiments with multiple sources of livestock exposure, the uncertainty regarding bioavailability across livestock exposure pathways is reduced or eliminated.

Chicken Dietary Intake

HHRAP recommends an overall chicken consumption rate of 200 g DW/day and assumes that this is composed entirely of grain, but this assumption is relevant to broiler chickens. It also recommends a soil consumption rate of 22 g/day. Laying hens consume less than broiler chickens, generally 100-150 g DW/day (Alabama A&M & Auburn Universities Extension, 2022). For this analysis, forage, drinking water, and homegrown hay are relevant exposure sources in the pasture farm scenario, with grain assumed to be from an uncontaminated, off-site source. Grain for chicken feed was assumed to be purchased from an uncontaminated source because purchasing feed is more common rather than growing it locally, likely due to the specific dietary needs of laying hens (Poultry Extension, 2024). The assumption is further supported by the finding that the grains typically included in poultry feed (oats, cracked corn) typically have low PFOA and PFOS accumulation (see Section 2.9.3.4). Because HHRAP assumes that chickens only consume grain, additional data sources for characterizing chicken diets were also sought. The EPA identified three studies or reports that included information about chicken dietary intakes.

Kowalczyk et al. 2020

This study on PFAS uptake by chickens and distribution to eggs included 12 hens fed a combination of highly contaminated hay (harvested from a field that received contaminated biosolids and paper-derived compost in southern Germany) and barley for 25 days. Kowalczyk et al. provided a detailed breakdown of the experimental chicken diet: 37% barley (grain), 8% hay, soybean meal (19%), triticale (28%), oil (1.5%), mineral feed (3%), and calcium carbonate (3.5%). The hens in this experiment were caged, so no opportunity for consumption of soil or insects occurred.

Dal Bosco et al. 2014

This study analyzed the impact of range enrichment (either sorghum plantings or olive trees) on behavior and diet of 250 free-range broiler chickens in each of two seasons on two farms (1,000 birds total). Forage intake was calculated for five subareas at increasing distance from the shelter. Total forage intake per bird (summed across the different distances from the shelter) for the sorghum-planted ranges are 30 g dry weight (DW)/day in summer and 18 g DW/day in winter. For the unenriched ranges, the corresponding values reported are 15 g DW/day for both seasons. However, the authors note that forage intakes for laying hens are likely higher, due to the younger age of broiler chickens compared to laying chickens. The authors cite several other studies that found values for laying hens in the 30–40 g DW/day range.

RAAF Base Williamtown, Australia site investigation (AECOM 2017)

The Australian Department of Defense completed an investigation and risk assessment associated with PFAS contamination around a base that used PFAS-containing firefighting foam. The risk assessment included a commissioned study of PFOA and PFOS uptake into chicken eggs (Wilson et al., 2020). Additional supplemental information from that study was published in a Department of Defense report (AECOM, 2017). This report includes a water intake for chickens of 0.208 L/day.

Diet Selection for Chickens

Starting with the total diet of 200 g DW/day from HHRAP, the diet fractions for silage and grain from Kowalczyk were applied to obtain intakes of 16 g DW/day of hay and 74 g DW/day of grain. For forage, a value of 30 g DW/day from Dal Bosco was selected. For soil ingestion, the value from HHRAP was rounded to 20 g/day and used. Finally, the water consumption rate from the Australian Department of Defense report of 0.208 L/day was rounded to 0.21 L/day and used (AECOM, 2017). These values are summarized in Appendix B.

Cow Dietary Intake

HHRAP recommends cattle consumption rates for forage, silage, grain, and soil for both beef and dairy cattle. For dairy cows, the dietary intake rates are 13.2 kg DW/day forage, 4.1 kg DW/day silage, and 3.0 kg DW/day grain. Water consumption rates for cows vary according to many factors, such as breed, body size, amount of milk produced per day, air temperature, humidity, and moisture content of feed (Harris and Van Horn, 1992). An analysis of water intake rates done for the 3MRA modeling system (US EPA, 2003g) was used to select a water intake of 92 L/day. That value reflects the average of data measured by Harris and Van Horn (1992) and reflect the variability in water consumption of dairy cows across different temperatures and milk production rates. This value falls within the water consumption ranges reported for other cow breeds by the University of Nebraska Cooperative Extension (1998), which for lactating Holstein cows was reported as 18 to 40 gallons/day, equivalent to 68–151 L/day. For beef cattle, the dietary intake rates are 8.8 kg DW/day forage, 2.5 kg DW/day silage, 0.47 kg DW/day grain, and 0.5 kg DW/day soil. The drinking water intake rate for beef cattle is 53 L/day (US EPA, 2003g).

2.9.3.7 Bioaccumulation Factors in Fish

The EPA selected fish bioaccumulation factors for this biosolids risk assessment to be consistent with draft AWQC for the Protection of Human Health for PFOA and PFOS (US EPA, 2024o,p). The EPA calculated draft BAFs for the PFOS and PFOA human health AWQC based on each chemical's properties (*e.g*., ionization and hydrophobicity), metabolism, and biomagnification potential (US EPA, 2024o,p; US EPA, 2000a; 2003h). The EPA's national BAFs represent the long-term, average bioaccumulation potential of a chemical in aquatic organisms that are commonly consumed by humans throughout the United States (US EPA, 2000a). The EPA evaluated results from field BAF and laboratory BCF studies on aquatic organisms commonly consumed by humans in the United States for use in developing national trophic-level BAFs.

To develop the draft BAFs for PFOA and PFOS, the EPA conducted a systematic literature search in October 2022 of publicly available literature sources to determine whether they contained information relevant to calculating national BAFs for human health AWQC (US EPA, 2000a; 2003h). The literature search for reporting the bioaccumulation of PFOA and PFOS was implemented by developing a series of chemical-based search terms, consistent to the process used in the derivation of BAFs used in the development of the Final Aquatic Life AWQC for PFOA and PFOS (US EPA, 2024l;m) and described in

Burkhard (2021). These terms included chemical names and CAS numbers, synonyms, tradenames, and other relevant chemical forms (*i.e.*, related compounds). Databases searched were Current Contents, ProQuest CSA, Dissertation Abstracts, Science Direct, Agricola, TOXNET, and UNIFY (database internal to the EPA's ECOTOX database). The literature search (including literature published through the first two quarters of 2020) yielded >37,000 citations that were further refined by excluding citations on analytical methods, human health, terrestrial organisms, bacteria, and where PFOA or PFOS was not a chemical of study. The citations meeting the search criteria were reviewed for reported BAFs and/or reported concentrations in which BAFs could be calculated. Data from papers that met the inclusion and data quality screening criteria described below were extracted into the chemical dataset.

Specifically, studies were evaluated for inclusion in the dataset used for calculating national BAFs using the following evaluation criteria:

- Only BAF studies that included units for tissue, water, and/or BAFs were included.
- Mesocosm, microcosm, and model ecosystem studies were not selected for use in calculating BAFs.
- BAF studies in which concentrations in tissue and/or water were below the minimum level of detection were excluded.
- Only studies performed using freshwater or brackish water were included; high salinity values were excluded.
- Studies of organisms (e.g., damselfly, goby) and tissues (e.g., fish bladder) not commonly consumed by humans or not used as surrogate species for those commonly consumed by humans were excluded. Information on the ecology, physiology, and biology of the organism was used to determine whether an organism is a reasonable surrogate of a commonly consumed organisms.
- Studies in which the BAFs were not found to be at steady state were excluded.
- For pooled samples, averaging BAF data from multiple locations was only considered acceptable if corresponding tissue and water concentrations were available from matching locations (e.g., a BAF would not have been calculated using water and tissue samples collected from eight separate locations with tissue concentrations collected from only six of these corresponding locations).

In addition to the evaluation criteria listed above, PFOS bioaccumulation data were also evaluated using five study quality criteria outlined in Burkhard (2021) and shown in Table 15.

Table 15. Study Quality Criteria Used by Burkhard (2021)

Criteria	1	2	3
Number of water samples collected	> 3 samples	2–3 samples	1 sample
Number of organism samples collected	> 3 samples	2–3 samples	1 sample
Temporal coordination of water and biota samples	Concurrent collection of samples	Collected within a 1 year time frame	Collected > 1 year time frame
Spatial coordination of water and biota samples	Collected from same locations	Collected from reasonably close locations (1 kilometer (km)–2 km)	Significantly different sampling locations
General experimental design	Assigned a default value of zero for studies in which tissues from individual species were identified and analyzed		Assigned a value of 3 for studies in which tissues were from mixed species or reported as a taxonomic group.

Note: The scores for each BAF were totaled and used to determine the overall confidence ranking for each individual BAF. The sum of quality values for the five criteria listed in Table 2 were classified as high quality (total score of 4 or 5), medium quality (total score of 5 or 6) or low quality (total score ≥ 7). Only high and medium quality data were included in final national BAFs calculations.

For the detailed derivation of PFOA and PFOS national BAFs, see US EPA 2024o and US EPA 2024p. Table 16 summarizes the draft national BAFs for PFOA and PFOS for trophic levels 3 and 4. Trophic level 2 BAFs are not relevant to the fish consumption scenarios assessed in this document (see Section 2.9.3.8, Fish Consumption Rate).

Table 16. Fish BAFs by Trophic Level

Trophic Level	PFOA (L/kg)	PFOS (L/kg)
TL3	49	1,700
TL4	31	860

2.9.3.8 Consumption Rates for Food and Water

The exposure factors used to parameterize the central tendency approach are selected to represent median values for the distribution of people represented by the various receptors captured in the conceptual models. If median values are not available, a mean value is used instead. Most of the exposure parameters are selected from tables presented in the most recent version of EPA's EFH; unless otherwise noted, that is US EPA (2011). Note that the exposure factors used for the central tendency modeling run are not those that would be used to calculate a risk-based regulatory threshold. A summary of human exposure factors can be found in Appendix B, Table B.12.

Fish Consumption Rate

In this assessment, the EPA selected a fish consumption rate of 0.47 g/kg-day for adults (~1.3 ounces per day), 0.31 g/kg-day for children 12-19 (~0.6 ounces per day), and 0.55 g/kg-day for children 6-11 (~0.6 ounces per day). These values represent the 50th percentile of Consumer-Only Intake of Home-Caught Fish (EFH Chapter 13, table 13-20). This survey did not have sufficient sample size to calculate fish intake rates for children aged 1-5, so the intake rate for children aged 6-11 was used for this group. A typical fish meal for adults is a 4 ounce to ½ pound serving of raw fish, which is 113-227 g; the adult fish consumption rate used in this assessment (assuming an 80 kg adult bodyweight) amounts to consuming an average of one to two fish servings per week. Serving sizes for children increase from 1 to 4 oz as they age from 1 to 11 years old. These exposure factors also equate to about 1-2 meals per week for children aged 1-5 and 6-11. Assuming a serving size of 4 oz per fish meal, the intake rate for children aged 12-19 equates to about one meal per week.

Bioaccumulation rates for PFOA and PFOS differ by trophic level (see Section 2.9.3.7). In this assessment, fish consumption is apportioned between trophic level 3 and trophic level 4 using data presented in EFH Chapter 10, Table 10-74, Total Consumption of Freshwater Fish Caught by All Survey Respondents During the 1990 Season. The species presented in this table were assigned trophic levels from the three following sources, in order of preference: 1) Estimated Fish Consumption Rates for the US Population and Selected Subpopulations (NHANES 2003-2010), Table 3 (US EPA, 2014); 2) The journal publication "Comparing trophic position of freshwater fish calculated using stable nitrogen isotope ratios (δN15) and literature dietary data," (Zanden et al., 1997); and 3) A publicly available database that catalogues information on various fish species published in the Journal of Fish Biology, Journal of Applied Ichthyology, and Acta Ichthyologica et Piscatoria (FishBase, 2024). The survey data presented in EFH Table 10-74 indicate that 14% of freshwater fish consumption is of fish in trophic level 3 (for example, lake whitefish, chub), while 86% of fish consumption is of fish in trophic level 4 (for example, brown trout, yellow perch, smallmouth bass).

Drinking Water Intake Rate

The drinking water intake rates for the central tendency modeling effort were selected from the latest edition of EPA's EFH, chapter 3 (ingestion of water and other select liquids; US EPA, 2019c). The values selected represent the 50th percentile of reported direct and indirect consumption of community water, in milliliters per bodyweight per day from the NHANES 2005-2010 survey (US EPA, 2019c, Table 3-21). The median drinking water intake is 13.4 ml/kg-day for adults, 6.5 ml/kg-day for children 12-19, 11.5 ml/kg-day for children 6-11 and 16.2 ml/kg-day for children 1-5. Assuming a bodyweight of 80 kg, this amounts to an adult drinking water intake rate of approximately 1 L/day.

Note that this drinking water intake rate used in this central-tendency modeling run is significantly lower than the drinking water intake rate used for other CWA purposes, such as development of national recommended human health criteria, and for Safe Drinking Water Act purposes, such as developing regulatory standards or setting non-regulatory health advisories.

Protected Fruits and Vegetables Intake Rates

"Protected produce" is a fruit or vegetable that has an outer protective coating that is typically removed before consumption. Examples of protected vegetables included pumpkin, corn, peas, and beans. Examples of protected fruits include melons like watermelon and cantaloupe, citrus fruits like oranges and grapefruit, and bananas.

The intake rates for protected fruits are the 50th percentile values in EFH chapter 13, table 13-59 and are presented in grams wet-weight fruit per kilogram of bodyweight per day. The median consumption rate of protected fruit is 2.1 g/kg-day for adults, 1.2 g/kg-day for children 12-19, 2.3 g/kg-day for children 6-11 and 2.3 g/kg-day for children 1-5. Given that a typical serving of fruit is 100-200 grams, the adult protected fruit intake equates to about one serving of protected fruit a day.

The intake rates for protected vegetables are the 50th percentile values in EFH chapter 13, table 13-61 and are presented in grams wet-weight vegetable per kilogram of bodyweight per day. The median consumption rate of protected vegetables is 0.6 g/kg-day for adults, 0.58 g/kg-day for children 12-19, 0.79 g/kg-day for children 6-11 and 1.4 g/kg-day for children 1-5. Given that a typical serving of vegetables is about 100 grams, the adult protected vegetable intake equates to about one serving of protected vegetables every other day.

Unprotected Fruits and Vegetables Intake Rates

"Unprotected" or "exposed" foods are those that are grown above ground and may be contaminated by pollutants deposited on surfaces of the foods that are eaten. Examples of unprotected vegetables include cauliflower, tomatoes, eggplant, cucumber, snap peas, herbs, and mushrooms. Examples of unprotected fruits include fresh or dried apples, pears, peaches, grapes, and berries.

The intake rates for unprotected fruits are the 50th percentile values in EFH chapter 13, table 13-58 and are presented in grams wet-weight fruit per kilogram of bodyweight per day. The median consumption rate of unprotected fruit is 1.3 g/kg-day for adults, 0.61 g/kg-day for children 12-19, 1.11 g/kg-day for children 6-11 and 1.82 g/kg-day for children 1-5. Given that a typical apple is about 240 grams, the adult unprotected fruit intake equates to about one apple every other day.

The intake rates for unprotected vegetables are the 50th percentile values in EFH chapter 13, table 13-60 and are presented in grams wet-weight vegetable per kilogram of bodyweight per day. The median consumption rate of unprotected vegetables is 1.4 g/kg-day for adults, 0.66 g/kg-day for children 12-19, 0.64 g/kg-day for children 6-11 and 1.5 g/kg-day for children 1-5. Given that a typical serving of vegetables is about 100 grams, the adult unprotected vegetable intake equates to about one serving of unprotected vegetables every day.

Root Vegetables Intake Rates

Root vegetables are vegetables where the consumed portion of the plant is the root. Root vegetables often have different uptake rates of environmental contaminants than vegetables where other portions (stems, leaves) of the plant are consumed. Examples of root vegetables include onions, carrots, beets, turnips, and potatoes. The intake rates for root vegetables are the 50th percentile values in EFH chapter 13, table 13-62 and are presented in grams wet-weight fruit per kilogram of bodyweight per day. The median consumption rate of root vegetables is 0.88 g/kg-day for adults, 0.57 g/kg-day for children 12-19, 0.52 g/kg-day for children 6-11 and 0.69 g/kg-day for children 1-5. Given that a typical serving of vegetables is about 100 grams (~ ½ an average-sized russet potato), the adult root vegetable intake equates to about five servings of root vegetables a week.

Milk and Dairy Intake Rates

The milk consumption rates for the central tendency scenario models were selected from the most recent edition of the EFH chapter 11, Meats, Dairy Products, and Fat (US EPA, 2018b) and chapter 13, Home Produced Foods (US EPA, 2011). Although chapter 13 (Intake of Home-Produced Foods) included some national data on intake of milk and other dairy products, there was only one age category in the available surveys with sufficient sample size to calculate descriptive statistics (ages 20-39). The respondents were additionally divided between families that answer yes to the question "Did anyone in the household produce any animal products such as milk, eggs, meat, or poultry for home use in your household?" (described as "households who farm") versus families that answer yes to the question "Did anyone in the household operate a farm or ranch?" (described as "households who raise animals"). Because the description of "households who farm" was best aligned with the conceptual model for the pasture farm, the 50th percentile dairy intake from this survey was used for adults (12.1 g/kg-day). For an 80 kg adult, this amounts to approximately 34 fluid ounces of milk consumed per day, which is four, 8oz glasses.

Because there were no data available for milk consumption in children specifically from families that produce milk at home, national milk consumption data was used for these age categories. Note that this national data likely underestimates the amount of milk consumed by children who grow up on dairy farms. For example, the available data for milk intake in adults finds that adults who live on farms consume about six times more dairy than in adults in national surveys. The values selected for the child age categories represent the 50th percentile of reported dairy consumption rate, in grams wet weight per kilogram bodyweight per day from the NHANES 2005-2010 survey (US EPA 2018b, Table 11-4). The median milk intake is 4.3 g/kg-day for children 12-19 (amounts to ~1, 8 oz glass per day), 12 g/kg-day for children 6-11 (amounts to 1.5, 8 oz glasses a day), and 30 g/kg-day for children 1-5 (amounts to ~2, 8 oz glasses per day).

Beef Intake Rates

Beef consumption rates were selected from the EFH chapter 13, Home produced foods. The values selected represent the 50th percentile of reported beef consumption rate for consumers-only, in grams per kilogram bodyweight per day from the Nationwide Food Consumption Survey (NFCS), 1987-1988 (Table 13-33). The median beef intake is 1.6 g/kg-day for adults, which represents the median intake of respondents in "households who farm." The beef intake rate is 1.5 g/kg-day for children 12-19 (~3 ounces per day) and 2.1 g/kg-day for children 6-11 (~2 ounces per day). This survey did not have data available for beef intake for children 1-5. The models assume that the intake rate for this group is the same as the intake rate for the slightly older children of 2.1 g/kg-day. This assumption is supported by information provided in Chapter 11 of the EFH (meat, dairy, and fats; US EPA, 2018b), which reports in Table 11-6 that the mean beef intake rate for age 2-6 ranges from 1.6 to 1.7 g/kg-day for the general public (NHANES 2005-2010). Assuming a bodyweight of 80 kg, the adult consumption rate amounts to an adult beef intake rate of slightly over one, three ounce serving of beef every day.

Egg Intake Rates

Egg consumption rates were selected from the EFH chapter 13, Home Produced Foods. The value selected represents the 50th percentile of reported egg consumption rate for consumers-only in "households who farm," reported in grams per kilogram bodyweight per day from the NFCS, 1987-1988 (Table 13-40). The median egg intake is 0.7 g/kg-day for all ages. This survey does not include age breakdowns for children and adults. EFH chapter 11, Meat and Dairy (US EPA, 2018b) does not include a survey specific to egg consumption. Because of this lack of age-specific intake rates, the "all ages" value from Table 13-40 will be used to represent egg intake rates for all age groups. Given an adult bodyweight of 80kg and a 50g average egg mass, this amounts to an intake rate of about 1 egg per day.

Chicken Intake Rates

The chicken consumption rates for the central tendency scenario models were selected from EFH chapter 11, Meats, Dairy Products, and Fat (US EPA, 2018b) and chapter 13, Home Produced Foods (US EPA, 2011). Although chapter 13, Intake of Home-Produced Foods, included some national data on intake of poultry, there were limited age categories with sufficient sample size to calculate descriptive statistics (see Table 13-52, Consumer-Only Intake of Home-Produced Poultry). As described previously, the respondents categorized as from "households who farm" was best aligned with the conceptual model for the pasture farm, the 50th percentile poultry intake from this survey was used for adults (1.1 g/kg-day). This survey does not include chicken-specific consumption rates, but rather consumption rates for "poultry," which includes chicken, turkey, and other poultry. The EPA finds that this represents the best available data for parameterizing intake rates of home-produced chickens. For an 80 kg adult, this intake rate amounts to about one three-ounce serving of chicken every day.

Because there was no data available for chicken consumption in children specifically from families that produce their own food, national chicken consumption data was used for these age categories (EFH, chapter 11, Table 11-6). The survey available for this consumption category reports mean intake values rather than median intake values. Mean intake values are likely slightly higher than median intake values but are still appropriate for this central tendency modeling exercise. The survey also only reported intake rates for poultry, rather than chicken only. A separate survey (represented in chapter 11, Table 11-7; US EPA, 2018b) indicates that for most Americans, the majority of poultry intake is chicken. The values selected for the child age categories represent the mean reported poultry consumption rate, in grams wet weight per kilogram bodyweight per day from the NHANES 2005-2010 survey (US EPA 2018b, Table 11-6). The mean intake is 1.1 g/kg-day for children 12-19 (~2 oz per day), 1.6 g/kg-day for children 6-11 (~1.6 oz per day) and 2.4 g/kg-day for children 1-5 (~1.3 oz/day).

Overview of Consumption Rates

Table 17. Overview of Selected Human Consumption Rates

Category	Adult (g/kg-day for all except drinking water)	Child 1-5 (g/kg-day for all except drinking water)	Child 6-11 (g/kg-day for all except drinking water)	Child 12-19 (g/kg-day for all except drinking water)
Fish	0.47 (1.3 oz per day; ~1-2 servings a week)	0.55 (0.3 oz per day; ~1-2 servings a week)	0.55 (0.6 oz per day; ~1-2 servings a week)	0.31 g (0.6 oz per day; ~1 serving a week)
Drinking water	13.4 ml/kg-day (1 L per day)	16.2 ml/kg-day (240 ml per day)	11.5 ml/kg-day (330 ml per day)	6.5 ml/kg-day (300 ml/day)
Protected fruits	2.1 (6 oz per day; ~1 serving per day)	2.3 (1 oz per day)	2.3 (2.4 oz per day)	1.2 (2.6 oz per day)
Protected vegetables	0.6 (1.7 oz per day; ~½ serving a day)	1.4 (0.8 oz per day)	0.79 (0.8 oz per day)	0.58 (1.2 oz per day)
Unprotected fruits	1.3 (3.6 oz per day; ~1/2 an apple a day)	1.82 (1 oz per day)	1.11 (1.1 oz per day)	0.61 (1.3 oz per day)
Unprotected vegetables	1.4 (4 oz per day; ~ 1 serving per day)	0.64 (0.3 oz per day)	0.64 (0.65 oz per day)	0.66 (1.4 oz per day)
Root vegetables	0.88 (2.5 oz per day; ~ ½ a small potato a day)	0.69 (0.4 oz per day)	0.52 (0.5 oz per day)	0.57 (1.2 oz per day)
Milk and dairy	12.1 (34 oz a day; ~4, 8 oz glasses)	30 (15 oz per day; ~2, 8 oz glasses per day)	12 (12 oz per day; ~1.5, 8 oz glasses a day)	4.3 (9 oz per day; ~1, 8 oz glass per day)
Beef	1.6 (4.5 oz per day)	2.1 (1 oz per day)	2.1 (2.1 ounces per day)	1.5 (3.2 ounces per day)
Egg	0.7 (~1 egg per day)	0.7 (~1 egg every 5 days)	0.7 (~1 egg every other day)	0.7 (~1 egg per day)
Chicken	1.1 (3.1 oz per day, ~1 serving per day)	2.4 (1.3 oz/day)	1.6 (1.6 oz per day)	1.1 (2 oz per day)

2.9.3.9 Cooking and Food Preparation Loss Assumptions

Risk assessments that include food consumption pathways often consider if a portion of the contaminant is lost during the food prep or cooking process. EFSA conducted an assessment of ingestion risks for PFOA and PFOS through food exposures in 2018 (EFSA CONTAM Panel, 2018). In the assessment, the authors summarized the available literature on food loss in preparing or cooking various types of food containing PFOA and PFOS. They find that some studies report loss of PFOA and PFOS while other studies find PFOA and PFOS concentrations increase, perhaps due to loss of water during the cooking process, which increases the concentration of remaining contaminant. Overall, ESFA concludes that the limited number of studies gives an inconsistent view about whether losses or increases occur for PFOA and PFOS across different food types and cooking strategies. The biosolids draft risk assessment will thus assume 0% loss in fruits, vegetables, meats, eggs, and milk.

2.9.3.10 Soil Ingestion Rates

The soil ingestion rates for the central tendency modeling effort were selected from the EFH, chapter 5 (soil and dust ingestion). The values selected represent the central tendency of soil ingestion (which includes soil and outdoor dust), in mg per day (US EPA 2017, Table 5-1). The central tendency soil ingestion rate is 10 mg/day for adults, 10 mg/day for children 12-19, 30 mg/day for children 6-11 and 40 mg/day for children 1-5. The EFH notes that soil and dust ingestion is likely higher in adults following a "traditional rural or wilderness lifestyle." It is likely that this central tendency estimate would underestimate soil ingestion for a farmer who frequently works weeding, harvesting, or otherwise disturbing soils on a farm. However, the EFH does not include a dust ingestion rate specific for adults who work on farms.

2.9.3.11 Body Weight

In this draft risk assessment, the EPA selected a bodyweight of 80 kg for adults, 61 kg for children 12-19, 29 kg for children 6-11 and 15 kg for children 1-5. These rates are based on 50th percentile American bodyweight, Table 8-3 of US EPA (2011), NHANES 1990-2006. Note that bodyweight assumptions are only required when bodyweight-normalized intake rates are not available.

2.9.3.12 Duration of Exposure Modeling

The exposure model does not assume that the residents spend their entire life at the relevant site; rather, it is assumed that the residents have moved over the course of their life. For this draft risk assessment, the EPA selected an exposure duration of 10 years for adults, corresponding to the 50th percentile of total residence time for farms from Table 16-113 of US EPA (2011). The 50th percentile of residential occupancy from the EFH, Table 16-109 is 9 years. Thus, 10 years is a reasonable value for nearby residents who are not farmers as well. This residency assumption applies to the entire family, including children. The exposure period for cancer risk and non-cancer is assumed to occur around the time of maximum media concentrations within the modeling period (so, if the peak media concentration occurs in model year 40, the 10-year exposure duration would run from model years 35 to 44 and the 1-year exposure duration would be for model year 40). The cancer exposure model assumes that the receptors are at the relevant site for 350 days per year (either their non-farm home or farm home, depending on the conceptual model).

Because an exposure duration of 10 years is used for the entire farm family, the exposure factors for children aged 1-5 and 6-11 were combined (using a weighted average based on sample size in each age bin reported) into values appropriate for ages 1-11. For soil consumption, which is not based on a single study, this assessment used the slightly higher value of 40 g/day (for children 1-5, vs 30 g/day for children 6-11) for children 1-11. These average intake values are provided in Appendix B.

2.9.3.13 Location-specific Parameters

Models were parametrized to represent a range of climatological conditions (dry, moderate, and wet) using datasets from three regions located near Boulder, CO; Chicago, IL; and Charleston, SC. These locations were used as a basis for selecting, in order of preference, representative local (*e.g.*, meteorological parameters), regional (*e.g.*, soil and hydrologic parameters), and national data (*e.g.*, application characteristics). Where distributions of parameter values are available at the regional or national level, median values were selected.

Meteorological data. Daily meteorological data (precipitation, temperature) from a five-mile radius surrounding the three locations were represented by the nearest gridded dataset developed by the EPA primarily for pesticides modeling (Fry et al., 2016). The mean annual windspeed for each region was also identified. Parameters describing general soil properties in the field and surrounding watershed for overland flow and transport calculations are represented by median values selected from national distributions developed in support of other pollutant evaluations for the EPA (see Table B-6). By selecting the weather and soil data from the same geographic location, the models are pairing climate and soil conditions that naturally co-occur.

Application location size. Parameters describing general site characteristics applicable to crop, pasture, and reclamation land application scenarios are also based on median national values developed as part of various Federal agency missions (*e.g.*, USDA national farm field sizes) or in support of other pollutant evaluations for EPA. The 80-acre field[14] where biosolids are applied, and which is used to grow crops (crop scenario) or to pasture cows (pasture and reclamation scenarios), is assumed to be square. Though the model allows for the site to have vehicles and corresponding particulate spread through dust, this assessment assumes no vehicles regularly drive over the site.

Surface water size, location. A 13-acre index reservoir[15] that drains the adjacent local watershed serves as an alternative source of drinking water for the farm family (their primary drinking water source is assumed to be groundwater). The index reservoir is based on the standard waterbody parameters for VVWM, the waterbody model used to estimate concentrations in surface water (US EPA, 2019b; 2020). A 10-meter wide, rectangular buffer exists between the field and the index reservoir;[16] the LAU source model estimates runoff and erosion from the field to the buffer and then from the buffer to the reservoir. The farm family is assumed to live in the buffer.

Soil characteristics. Soil characteristics for determining regional recharge rates to groundwater and to parameterize the unsaturated portion of the groundwater model are based on the predominant soil mega-texture within a 5-mile radius of the field location from the same national data source as the watershed characteristics. The EPA HELP model (Schroeder et al., 1994) was used to calculate regional recharge rates using meteorological data assigned to each location, and HELP default values for the following parameters corresponding to predominate soil mega-texture at each location:

- *Soil Porosity*: ratio of the volume of void spaces in a volume of soil.
- *Field Capacit*y: The volume of water remaining in void spaces in a volume of soil after freely draining from a saturated state, expressed as a percentage.
- *Wilting Poin*t: volume of water remaining in void spaces in a volume of soil at which plants wilt and fail to recover, expressed as a percentage.
- *Soil Hydraulic Conductivit*y: the amount of water moving vertically through a unit area of saturated soil in unit time under unit hydraulic gradient.

The following parameters are used in EPACMTP to describe flow in the unsaturated zone in addition to the soil hydraulic conductivity:

- *Alpha and Beta*: soil-specific shape parameters used in the van Genuchten (1980) model for modeling soil-water content as a function of pressure head.
- *Residual water conten*t: the irreducible water content obtained after lowering the pressure head in the soil.
- *Saturated water conten*t: maximum fraction of total volume of soil occupied by water in the soil (equivalent to soil porosity).
- *Percent Organic Matte*r: measure of amount of organic material present within the soil of the unsaturated zone, as a weight percent.

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[14] The field size is based on the 50th percentile from the *2012 Census of Agriculture* (USDA, 2014).

[15] The index reservoir is based on the standard waterbody parameters for Variable Volume Water Model (VVWM), the waterbody model used to estimate concentrations in surface water (US EPA, 2019; 2020); see Section A.2.3.2.

[16] The Part 503 regulations state that "bulk sewage sludge shall not be applied to agricultural land, forest, or a reclamation site that is 10 meters or less from waters of the United States." The buffer for the index reservoir has been set to 10 m in accordance with this standard.

The values for each of these parameters are based on median values specific to each mega-texture associated with each location selected from national distributions developed in support of other pollutant evaluations for EPA.

Aquifer characteristics. Aquifer characteristics (depth to water table, aquifer thickness, regional hydraulic gradient, and aquifer hydraulic conductivity) were based on median values from the EPA's Hydrogeologic Database (HGDB). The HGDB was developed by the American Petroleum Institute (Newell et al., 1989; 1990) to specify correlated data sets of these four parameters for the 12 distinct hydrogeologic environments described in Newell et al. (1990). The EPA first developed a national geographic coverage of the 12 hydrogeologic environments, and then used GIS to overlay the three simulated locations and assign each location a hydrogeologic environment. Median values were selected for each of the four parameters from the assigned environments. One exception was at the "wet" region, where a mean value for the hydraulic conductivity of the saturated zone was used instead of the median. The use of the median value (315 m/yr), in conjunction with other inputs, resulted in a mounded water table that exceeded the elevation of the ground surface, violating an underlying assumption of EPACMTP model (Section 4.3.6 of EPA 2003e). Adjusted values for this parameter input are also noted in Appendix B. Other flow and transport-related parameters not associated with chemical properties are selected from national distributions developed in support of other pollutant evaluations for the EPA (US EPA, 2003a) or where specifically noted in Appendix B. These parameters include aquifer porosity, bulk density, dispersivity, aquifer fraction of organic content, and temperature.

2.9.3.14 Biosolids Application Assumptions

Biosolids applications of 10 MT dry weight per hectare of field area were modeled to occur once per year on April 1 for 40 years for the crop and pasture scenarios. The EPA's prior risk assessment of dioxins and PCBs also used a 40 year timeframe for application of sewage sludge to a field (US EPA, 2003a). The existing sewage sludge regulations in 40 CFR part 503 assume 100 consecutive years of sewage sludge land application when calculating cumulative and annual loading rates for metals. As there are not data available on the longevity of sewage sludge application to a given field or location, the EPA is continuing to model risks for scenarios with 40 years of application, in line with the prior risk assessment. To estimate a reasonable median agronomic application rate, probabilistic plant available nitrogen (PAN) calculations were conducted using the PAN and agronomic spreadsheet calculation tool available from the Colorado Department of Public Health & Environment (CDPHE, 2018) and @Risk (Palisade Corporation), a Microsoft Excel plug-in. The basic annual rate calculation is based on PAN per metric ton of biosolids on a per hectare basis and the crop nitrogen requirement. Probabilistic simulations were conducted assuming an absence of residual nitrogen from any sources (background or previous biosolids or fertilizer application) and varying several parameters such as crop yield and days to incorporation. The analysis is described in more detail in Appendix E of US EPA (2023c). This produced a range for dry weight agronomic application rate of approximately 0.5 to 30 dry MT/ha and an overall median value of 7.6 dry MT/ha. This range is consistent with recommended ranges found elsewhere in the literature for crop applications (US EPA, 2000b), which range from around 2 to 20 dry MT/ha. The application rate value of 10 dry MT/ha used in this assessment is based on rounding the analysis median value to the nearest order of magnitude to account for variability. Biosolids are assumed to be tilled (*i.e.*, fully mixed) into the top 20 cm of the field for the crop scenario whereas in pasture and reclamation scenarios, biosolids are assumed to be unincorporated with field soils after application.

For the reclamation scenario, a single application of biosolids at a rate of 50 MT dry weight per hectare of field area is modeled to occur on the April 1 of the first year of the simulation.

2.9.3.15 Surface Disposal Assumptions

The surface disposal unit (SDU) is modeled as having a square footprint with an area of 3,400 m^2; this value is calculated from the median values of depth and flow from the Industrial D Screening Survey data presented in the 3MRA modeling documentation (US EPA, 2003g) and the operating life described below. The SDU is assumed to operate for 50 years, consistent with the 2003 sewage sludge screening assessment (US EPA, 2003a; appendix G), during which time, liquids and dissolved chemical mass in the liquids can pass through the bottom of the unit. Liquids in the unit are assumed to maintain a near constant volume and are not aerated. Darcy's law is used to calculate the rate of leakage through the base of the unit into the unsaturated zone, and the base of the unit may be unlined, clay lined, composite lined. After 50 years, the SDU is assumed closed and no additional chemical mass is released to the environment; however, the groundwater model assumes the long-term average volumetric rate of liquids leaving the unit are assumed to persist to beyond 50 years. The source of groundwater for drinking is assumed to be 5 meters down gradient of the SDU, in the middle of a 10-meter buffer area (the same as the land application scenarios). The surface disposal unit is assumed to be "clean closed" at the end of its 50-year economic life such that no residual PFOA or PFOS remains.

The key parameters governing the rate of leakage through the bottom of the SDU are, as organized by liner scenario:

- Unlined and Clay Lined SDUs:
  - The maximum height of liquids above the bottom of the SDU (2 m)
  - Flow rate into the SDU: (4×10^{-6} m^3/s)
  - Precipitation rate that are specific to each of the three locations representing dry, average, and wet climates
  - Material properties of the settled sediment in the SDU, including saturated hydraulic conductivity (5×10^{-7} m/s) and soil-water retention parameters (Alpha 0.016 1/cm; Beta 1.37)
- Clay Lined SDUs
  - Material properties and dimensions of the clay liner, including saturated hydraulic conductivity (1×10^{-9} m/s), soil-water retention parameters (Alpha 0.008 1/cm; Beta 1.09), and liner thickness (0.9144 m)
- Composite Lined SDUs:
  - Specified infiltration rate through a composite liner (1.4×10^{-6} m/d)[17]

The key processes and non-chemical specific parameters governing the concentration of chemical mass of PFOA and PFOS in the liquids passing through the bottom of the SDU are limited to sorption and solids generation and removal:

- Influent total suspended solids concentration (0.1 g/cm^3)
- Fraction organic carbon in suspended solids (0.4 g/g)
- Solids removal rate (calculated based on flow rate, SDU area, and suspended particle sizes).

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[17] The approximate 90th percentile infiltration rate from Table 4.6 of US EPA, 2003d.

3 ANALYSIS

3.1 Exposure Characterization, Central Tendency Models

The following sections present and discuss the modeled concentration and exposure results for individual exposure pathways in each of the biosolids use or disposal scenarios outlined in Section 2.8. The modeled media concentration results are presented in units of ng PFOA or PFOS per mg wet weight of media (*e.g.*, milk, soil, water, beef). All modeling runs assume that the starting concentrations of PFOA and PFOS in sewage sludge are 1 ppb (1 µg/kg). This concentration is near available detection thresholds for PFOA and PFOS in sewage sludge (US EPA, 2024d) and below levels commonly detected in U.S. sewage sludge (see Section 2.4 and Appendix A). The models and calculations used in this assessment have a linear relationship between the starting concentration of PFOA and PFOS in sewage sludge and the modeled concentration in each environmental media. This means that if the starting concentration of PFOA or PFOS in sewage sludge were to increase from 1 ppb to 10 ppb, the modeled media concentration would increase by a factor of 10.

As described in Section 5.3, the concentration results from fate and transport modeling are highly sensitive to the parameters associated with the climate setting and K_{oc}. For this reason, modeled exposures for a given pathway will be presented for each climate (dry, moderate, and wet) and for a low K_{oc} (10th percentile) and high K_{oc} (90th percentile).

3.1.1 Crop Farm

The crop farm scenario models the fate and transport of PFOA and PFOS as they move from biosolids through soil, surface water, and groundwater. The models then estimate the direct exposure to adults and children to those media, and the uptake and exposure from those media to fruits, vegetables, and fish. In this central tendency modeling exercise, the concentrations of PFOA and PFOS in the modeled biosolids are low (1 ppb) for each chemical. The following tables show the modeled concentrations of PFOA and PFOS in each media type during either a ten-year averaging time or a one-year averaging time. These averaging windows include the maximum concentration year for each media type. The tables include three climate scenarios: dry, moderate, and wet. These climate scenarios also represent varied soil types, depths to groundwater, hydrological connectivity and other related hydrogeological conditions that would be expected in these climate settings.

Table 18. PFOA Media Concentrations for Crop Farm (ppt): Maximum 10- and 1-year Averages

Pathway	Low K_{oc}		High K_{oc}	
	10-yr	1-yr	10-yr	1-yr
Dry Climate				
Exposed Fruit	0.86	0.89	2.0	2.1
Exposed Vegetables	6.6	6.9	15	16
Fish[a]	260	270	48	49
Groundwater	0.40	0.45	2.1E-9	2.1E-9
Protected Fruit	0.74	0.77	1.7	1.8
Protected Vegetables	13	13	29	30
Root vegetable	6.1	6.4	14	15
Soil	34	43	92	100
Surface water	7.8	8.0	1.4	1.4
Moderate climate				
Exposed Fruit	0.050	0.076	0.81	0.86
Exposed Vegetables	0.39	0.58	6.2	6.6
Fish[a]	14	15	48	51
Groundwater	4.5	5.5	0.12	0.12
Protected Fruit	0.044	0.066	0.70	0.75
Protected Vegetables	0.74	1.1	12	13

Pathway	Low K_{oc}		High K_{oc}	
	10-yr	1-yr	10-yr	1-yr
Root vegetable	0.36	0.54	5.8	6.1
Soil	0.56	1.8	29	39
Surface water	0.42	0.46	1.4	1.5
Wet Climate				
Exposed Fruit	0.046	0.088	0.62	0.64
Exposed Vegetables	0.35	0.67	4.8	4.9
Fish[a]	9.5	12	34	36
Groundwater	4.5	4.5	0.48	0.48
Protected Fruit	0.04	0.076	0.54	0.56
Protected Vegetables	0.67	1.3	9.0	9.4
Root vegetable	0.33	0.62	4.4	4.6
Soil	0.52	1.6	21	27
Surface water	0.28	0.36	1.0	1.1

[a] These values represent the weighted average fish tissue concentration by the percent consumption of trophic levels 3 and 4.

Table 19. PFOS Media Concentrations for Crop Farm (ppt): Maximum 10- and 1-year Averages

Pathway	Low K_{oc}		High K_{oc}	
	10-yr	1-yr	10-yr	1-yr
Dry Climate				
Exposed Fruit	0.33	0.34	0.57	0.58
Exposed Vegetables	0.81	0.83	1.4	1.4
Fish[a]	4500	4500	40	41
Groundwater	0.054	0.059	4E-31	4E-31
Protected Fruit	0.29	0.29	0.49	0.5
Protected Vegetables	1.5	1.6	2.6	2.7
Root vegetable	11	11	19	20
Soil	60	70	120	130
Surface water	4.6	4.6	0.039	0.039
Moderate climate				
Exposed Fruit	0.076	0.083	0.50	0.53
Exposed Vegetables	0.19	0.2	1.2	1.3
Fish[a]	1700	1800	49	52
Groundwater	0.98	0.98	1.8E-05	1.8E-05
Protected Fruit	0.066	0.072	0.44	0.46
Protected Vegetables	0.35	0.38	2.3	2.5
Root vegetable	2.6	2.8	17	18
Soil	8.4	14	83	110
Surface water	1.7	1.8	0.048	0.051
Wet Climate				
Exposed Fruit	0.055	0.062	0.46	0.47
Exposed Vegetables	0.13	0.15	1.1	1.2
Fish[a]	1100	1100	54	57
Groundwater	2.7	2.7	0.01	0.015
Protected Fruit	0.048	0.054	0.4	0.41
Protected Vegetables	0.25	0.29	2.1	2.2
Root vegetable	1.9	2.1	15	16
Soil	4.9	8.9	84	97
Surface water	1.1	1.1	0.052	0.055

[a] These values represent the weighted average fish tissue concentration by the percent consumption of trophic levels 3 and 4.

The crop farm scenario outputs concentrations over time for two categories of fruits (exposed and protected), three categories of vegetables (exposed, protected, and root), fish, surface water, soil, and groundwater. Groundwater concentrations range from effectively 0 ng/L to 5.5 ng/L for PFOA and effectively 0 to 2.7 ng/L for PFOS. Surface water concentrations range from 0.028 to 8.0 ng/L for PFOA and from 0.039 to 4.6 ng/L for PFOS. Soil concentrations range from 0.52 to 100 ng/kg for PFOA and 4.9

to 130 ng/kg for PFOS. Fish tissue concentrations range from 9.5 to 270 ng/kg for PFOA and 40 to 4,500 ng/kg for PFOS. Finally, fruit and vegetable concentrations range from 0.040 to 30 ng/kg for PFOA and 0.048 to 20 ng/kg for PFOS. Root, protected, and exposed vegetables have higher PFOA and PFOS concentrations than the other produce categories.

Overall, the one-year and ten-year average concentrations for each media are similar. Ten-year average concentrations are often the same or only slightly lower than the one-year averages. This trend reflects the fact that for many media types, yearly average concentrations remain elevated for years at a time. See Section 3.2 for more discussion on temporal trends in modeled concentrations.

Potential groundwater contamination associated with PFOA and PFOS leaching from biosolids-amended soils is of high concern, in part because biosolids are often land-applied in areas where nearby residents rely on groundwater as a source of drinking water. This modeling exercise allows us to explore the potential impacts to groundwater at biosolids concentrations that are commonly exceeded around the U.S. (concentrations of 1 ppb for PFOA and PFOS) when they are annually applied to land used to grow fruits and/or vegetables. The draft modeling results show that when biosolids are applied with these low PFOA and PFOS concentrations, groundwater concentrations of PFOA and PFOS vary depending on the K_{oc} and climate setting in each modeled scenario. For PFOA, groundwater concentrations range from effectively zero in the high K_{oc} dry climate to 4.5–5.5 ng/L in the low K_{oc} moderate and wet climates. For PFOS, groundwater concentrations range from effectively zero in the high K_{oc} moderate and dry climate to 2.7 ng/L in the low K_{oc} wet climate setting. Overall, these groundwater results are similar to the results seen for the pasture farm scenario (see Section 3.1.2), which also models a farm setting, but the pasture farm scenario assumes no tilling of soil whereas the crop farm assumes annual tilling of the field.

These groundwater outcomes can be partially explained by the sorption behavior of PFOA and PFOS in soils. The K_d is calculated by measuring the concentration of PFOA or PFOS in soil and dividing it by the equilibrium concentration of PFOA or PFOS in the soil pore water. This metric indicates the relative amount of PFOA or PFOS that sorbs to soil in comparison to the amount dissolved in the surrounding water. In EPA's models, K_d is calculated by multiplying K_{oc} by the f_{oc} in the biosolids-amended soils for each climate setting (see Section 2.9.3.3). This allows the models to adjust K_d based on the amount of organic matter in the underlying soils for each climate and geological setting. However, measurements of K_d are more common than measurements of K_{oc} and more directly capture soil leaching potential in field conditions. PFOS generally has higher measured K_d than PFOA in biosolids-amended soils. Though observed K_d for both compounds in biosolids-amended soils can vary more than two orders of magnitude across locations, within a single study site, K_d values for PFOS are higher than those for PFOA. For example, in a recent study of PFOA and PFOS in biosolids-amended soils in New Hampshire, the average $\log(K_d)$ for PFOS was generally between 2 and 2.5 L/kg while the $\log(K_d)$ for PFOA was between 1 and 2 L/kg (Tokranov et al., 2023). Correspondingly, the model results show that a higher proportion of PFOS is retained in soils and a higher portion of PFOA is mobilized through the soil column to groundwater. These trends are reflected in both the soil and groundwater concentrations generated by modeling runs, in that when PFOA and PFOS are at the same concentration in biosolids (1 ppb), soil concentrations are higher for PFOS than PFOA while groundwater concentrations are higher for PFOA than PFOS. Note that when modeling the fate and transport of PFOA and PFOS from biosolids contaminated with concentrations of 1 ppb for each compound, the resulting groundwater concentrations are often, but not always, below the minimum reporting level (MRL) of 4 ng/L for each compound using EPA's groundwater method EPA 533.

Another media of high concern is fish tissue, especially for PFOS, which is known to be highly bioaccumulative in the commonly consumed portions of fish like filets. In this modeling scenario, PFOA and PFOS accumulate in fish after the chemicals leave the farm field and travel over a 10-meter soil buffer to the nearby surface water reservoir that is 13 acres in size. This transfer occurs in the models in the water phase through overland flow of dissolved and particle-bound mass, though PFOS or PFOA bound to particulates transported through the air could also be source to nearby waterbodies. The models include daily-scale meteorological data, which allows the model to capture episodic increases in runoff and erosion from storm events. This model does not include any connection between groundwater and the surface water reservoir. The surface water can be thought of as a source of drinking water or only as the route of PFOA and PFOS contamination to the fish. The concentration of PFOA and PFOS is linearly correlated to the size of the modeled surface water reservoir, such that if the volume of water in the reservoir increases by a given percentage, the concentration in PFOA and PFOS in surface water and fish tissue will decrease by the same percentage.

Overall, the daft modeling finds that surface water concentrations for PFOS are lower than surface water concentrations for PFOA across each climate and K_{oc} scenario. However, PFOS fish tissue concentrations are consistently higher than PFOA fish tissue concentrations in each scenario. This trend is due to the high BAFs for PFOS, which are 1,700 (trophic level 3) and 860 (trophic level 4), compared to the BAFs of 49 (trophic level 3) and 31 (trophic level 4) for PFOA. A recent FDA study using FDA's PFAS methods for food had a maximum residue level (MRL) of 39 ppt for PFOS and 90 ppt for PFOA (FDA, 2022). Modeled concentrations of PFOS in fish tissue are consistently above MRLs in the low K_{oc} scenarios, but not in the high K_{oc} scenarios, where more PFOS is retained in soil. The modeled concentrations of PFOA in fish tissue are consistently below MRLs in FDA's PFAS methods.

In some instances, surface water bodies are used for drinking water instead of groundwater. The results of the modeling exercises show that the concentrations of PFOA and PFOS in surface water are consistently higher than the concentrations in groundwater in a given modeling run. This indicates that those using a surface water reservoir as a source of drinking water would be expected to have higher PFOA and PFOS drinking water exposure than those using groundwater as a source of drinking water, assuming that biosolids are applied within ten meters of the reservoir. If biosolids were applied further from the drinking water reservoir or the reservoir were larger, the concentrations of PFOA and PFOS would decrease.

The PFOA and PFOS concentrations in fruits and vegetables predicted in these models are primarily dependent on 1) uptake factors for the grouping of plants, 2) modeled retention of PFOA and PFOS in the soils, and 3) the percent moisture factor used to convert dry weight to wet weight measurements. There are significant data limitations on the uptake factors used for each category of fruits and vegetables included in this assessment, which results in a high degree of uncertainty in the modeled plant concentrations. As described in Section 2.9.3.4, these limitations on available data for uptake factors in fruits and vegetables likely indicate that the exposures from fruits and vegetables are over-estimated. Given these limitations, there are some general trends that the modeling can show us. Though plant uptake factors are generally higher for PFOA than PFOS, more PFOS is generally retained in soils due to PFOS's higher K_{oc}. As a result, PFOA or PFOS concentrations can be higher in fruits and vegetables depending on the climate and K_{oc} setting. Exposed vegetables, where humans tend to eat leaves, shoots, or stalks (*i.e.*, spinach, celery, lettuce) tend to have the higher concentrations of PFOA and PFOS due to the higher uptake factors. Overall, the modeled concentrations for fruits and vegetables should be seen as rough estimates, with a high variability and uncertainty. Additional data of PFOA and PFOS uptake into fruits and vegetables, especially when these plants are grown on biosolids-impacted soils, would help reduce this uncertainty.

For reference, the exposures for each pathway for the crop farm are presented In Tables 20 and 21 in units of ng/kg-day. These exposures are calculated using the consumption rates described in Section

2.9.3.8 as well as other factors described in Sections 2.9.3.9 through 2.9.3.12. The lifetime average daily dose (LADD) averages the daily exposure during the exposure duration over a lifetime of 70 years and is used for calculating cancer risk (see Section 4.1, Equation 4). The average daily dose (ADD) averages the daily exposure over the exposure duration of one year, not the full lifetime, and is used for calculating noncancer hazard (see Section 4.1, Equation 5).

Table 20. PFOA Exposures for Crop Farm (ng/kg-day): LADD and ADD

	Low Koc				High Koc			
	Adult		Child		Adult		Child	
Pathway	LADD	ADD	LADD	ADD	LADD	ADD	LADD	ADD
Dry Climate								
Exposed fruit	0.00015	0.0012	0.00016	0.0012	0.00035	0.0027	0.00036	0.0028
Exposed vegetable	0.0013	0.0096	0.00090	0.0069	0.0029	0.022	0.0021	0.016
Fish	0.017	0.13	0.020	0.15	0.0031	0.023	0.0036	0.027
Groundwater	0.00073	0.006	0.00076	0.0062	3.8E-12	2.8E-11	3.9E-12	2.9E-11
Protected fruit	0.00021	0.0016	0.00023	0.0018	0.00049	0.0038	0.00054	0.0041
Protected vegetable	0.0010	0.0078	0.0019	0.014	0.0024	0.018	0.0043	0.033
Root vegetable	0.00073	0.0056	0.00049	0.0037	0.0017	0.013	0.0011	0.0087
Soil	5.8E-07	5.4E-06	8.8E-06	8.2E-05	1.6E-06	1.3E-05	2.4E-05	0.00019
Surface water	0.014	0.11	0.015	0.11	0.0026	0.019	0.0028	0.02
Moderate Climate								
Exposed fruit	9.00E-06	9.8E-05	9.2E-06	0.0001	0.00014	0.0011	0.00015	0.0011
Exposed vegetable	7.40E-05	0.00082	5.3E-5	0.00058	0.0012	0.0093	0.00085	0.0066
Fish	0.00092	0.0072	0.0011	0.0084	0.0031	0.024	0.0036	0.028
Groundwater	0.0082	0.074	0.0086	0.077	0.00023	0.0016	0.00024	0.0017
Protected fruit	1.3E-05	0.00014	1.4E-05	0.00015	0.00020	0.0016	0.00022	0.0017
Protected vegetable	6.1E-05	0.00066	0.00011	0.0012	0.00097	0.0076	0.0018	0.014
Root vegetable	4.3E-05	0.00047	2.9E-05	0.00032	0.00069	0.0054	0.00047	0.0036
Soil	9.6E-09	2.2E-07	1.5E-07	3.4E-06	5.0E-07	4.8E-06	7.6E-06	7.4E-05
Surface water	0.00078	0.0061	0.00081	0.0064	0.0026	0.02	0.0027	0.021
Wet Climate								
Exposed fruit	8.1E-06	0.00011	8.3E-06	0.00012	0.00011	0.00084	0.00011	0.00085
Exposed vegetable	6.8E-05	0.00094	4.8E-05	0.00067	0.00091	0.0069	0.00065	0.0049
Fish	0.00061	0.0056	0.00072	0.0066	0.0022	0.017	0.0026	0.02
Groundwater	0.0083	0.060	0.0086	0.063	0.00088	0.0064	0.00092	0.0067
Protected fruit	1.1E-05	0.00016	1.2E-05	0.00017	0.00015	0.0012	0.00017	0.0013
Protected vegetable	5.5E-05	0.00077	0.0001	0.0014	0.00074	0.0056	0.0014	0.01
Root vegetable	3.9E-05	0.00055	2.6E-05	0.00037	0.00053	0.004	0.00036	0.0027
Soil	8.9E-09	2.0E-07	1.4E-07	3.1E-06	3.6E-07	3.4E-06	5.4E-06	5.1E-05
Surface water	0.00052	0.0048	0.00055	0.005	0.0019	0.014	0.0019	0.015

Table 21. PFOS Exposures for Crop Farm (ng/kg-day): LADD and ADD

	Low Koc				High Koc			
	Adult		Child		Adult		Child	
Pathway	LADD	ADD	LADD	ADD	LADD	ADD	LADD	ADD
Dry Climate								
Exposed fruit	5.9E-05	0.00044	6.0E-05	0.00045	0.0001	0.00076	0.00010	0.00077
Exposed vegetable	0.00015	0.0012	0.00011	0.00083	0.00027	0.0020	0.00019	0.0014
Fish	0.29	2.1	0.34	2.5	0.0026	0.019	0.0030	0.022
Groundwater	9.9E-5	0.00078	0.00010	0.00082	7.4E-34	5.4E-33	7.7E-34	5.7E-33
Protected fruit	8.2E-05	0.00061	9.0E-05	0.00067	0.00014	0.0011	0.00015	0.0012
Protected vegetable	0.00013	0.00094	0.00023	0.0017	0.00022	0.0016	0.00040	0.0030
Root vegetable	0.0013	0.01	0.00090	0.0067	0.0023	0.017	0.0015	0.012
Soil	1.0E-06	8.8E-06	1.6E-05	0.00013	2.1E-06	1.6E-05	3.2E-05	0.00024
Surface water	0.0084	0.062	0.0087	0.065	7.2E-05	0.00053	7.5E-05	0.00055

Pathway	Low Koc				High Koc			
	Adult		Child		Adult		Child	
	LADD	ADD	LADD	ADD	LADD	ADD	LADD	ADD
Moderate Climate								
Exposed fruit	1.4E-05	0.00011	1.4E-05	0.00011	9.0E-05	0.00069	9.2E-05	0.0007
Exposed vegetable	3.6E-05	0.00028	2.5E-05	0.0002	0.00024	0.0018	0.00017	0.0013
Fish	0.11	0.83	0.13	0.97	0.0032	0.025	0.0037	0.029
Groundwater	0.0018	0.013	0.0019	0.014	3.2E-08	2.3E-07	3.4E-08	2.5E-07
Protected fruit	1.9E-05	0.00015	2.1E-05	0.00017	0.00013	0.00096	0.00014	0.0011
Protected vegetable	2.9E-05	0.00023	5.3E-05	0.00042	0.00019	0.0015	0.00035	0.0027
Root vegetable	0.00031	0.0025	0.00021	0.0016	0.0021	0.016	0.0014	0.011
Soil	1.4E-07	1.8E-06	2.2E-06	2.7E-05	1.4E-06	1.4E-5	2.2E-05	0.00021
Surface water	0.0032	0.024	0.0033	0.025	8.8E-5	0.00068	9.2E-05	0.00071
Wet Climate								
Exposed fruit	9.8E-06	8.1E-05	1.0E-05	8.3E-05	8.1E-05	0.00062	8.3E-05	0.00063
Exposed vegetable	2.6E-05	0.00021	1.8E-05	0.00015	0.00021	0.0016	0.00015	0.0012
Fish	0.068	0.52	0.079	0.61	0.0035	0.027	0.0041	0.031
Groundwater	0.0049	0.036	0.0051	0.037	1.9E-5	0.00020	2.0E-5	0.00021
Protected fruit	1.4E-05	0.00011	1.5E-05	0.00012	0.00011	0.00086	0.00012	0.00095
Protected vegetable	2.1E-05	0.00017	3.8E-05	0.00032	0.00017	0.0013	0.00032	0.0024
Root vegetable	0.00022	0.0019	0.00015	0.0012	0.0019	0.014	0.0012	0.0095
Soil	8.4E-08	1.1E-06	1.3E-06	1.7E-05	1.4E-06	1.2E-05	2.2E-05	0.00019
Surface water	0.0020	0.015	0.0021	0.016	9.6E-5	0.00074	0.0001	0.00077

3.1.2 Pasture Farm

The pasture farm scenario models the fate and transport of PFOA and PFOS as they move from biosolids through soil, surface water, and groundwater. The models then estimate the direct exposure to adults and children to those media, and the uptake and exposure from those media to animal feed, animal products, and fish. The pasture farm model includes the same assumptions about time living on the farm as the crop farm model. Notably, the pasture farm model does not include annual tilling of the farm fields, which is included in the crop farm model.

Table 22. PFOA Media Concentrations for Pasture Farm (ppt): Maximum 10- and 1-year Averages

Pathway	Low Koc		High Koc	
	10-yr	1-yr	10-yr	1-yr
Dry Climate				
Beef	5.2	7.7	31	32
Eggs	27	41	220	230
Fish[a]	340	340	140	140
Groundwater	2.8	2.8	0.026	0.026
Milk	8.4	12	44	46
Chicken	0.64	0.96	5.2	5.4
Soil	60	100	760	790
Surface Water	10	10	4.2	4.2
Moderate Climate				
Beef	1	1.3	4.3	5.7
Eggs	4.2	6.4	30	42
Fish[a]	60	64	49	52
Groundwater	4.3	4.3	0.27	0.27
Milk	1.7	2.1	6.2	8.3
Chicken	0.099	0.15	0.71	0.97
Soil	4.8	12	100	140
Surface Water	1.8	1.9	1.5	1.5
Wet Climate				
Beef	0.7	0.88	2.9	4.2
Eggs	2.9	4.2	20	30

Pathway	Low Koc		High Koc	
	10-yr	1-yr	10-yr	1-yr
Fish[a]	36	39	29	30
Groundwater	2.6	2.6	0.78	0.78
Milk	1.2	1.4	4.3	6.1
Chicken	0.067	0.098	0.47	0.69
Soil	3	7.8	65	97
Surface Water	1.1	1.2	0.86	0.88

[a] These values represent the weighted average fish tissue concentration by the percent consumption of trophic levels 3 and 4.

Table 23. PFOS Media Concentrations for Pasture Farm (ppt): Maximum 10- and 1-year Averages

Pathway	Low Koc		High Koc	
	10-yr	1-yr	10-yr	1-yr
Dry Climate				
Beef	120	140	280	290
Eggs	160	200	550	570
Fish[a]	8100	8300	240	240
Groundwater	0.22	0.22	6.1E-31	6.1E-31
Milk	20	24	40	41
Chicken	17	21	57	60
Soil	280	350	1100	1100
Surface Water	8.3	8.5	0.24	0.24
Moderate Climate				
Beef	29	33	170	180
Eggs	24	33	340	350
Fish[a]	2300	2500	230	230
Groundwater	1.1	1.1	6.8E-4	6.8E-4
Milk	5.1	5.7	25	26
Chicken	2.6	3.4	35	37
Soil	29	46	670	710
Surface Water	2.4	2.5	0.22	0.23
Wet Climate				
Beef	11	21	110	120
Eggs	13	22	220	230
Fish[a]	1300	1400	160	170
Groundwater	2	2	0.012	0.012
Milk	1.9	3.6	16	17
Chicken	1.4	2.3	23	24
Soil	21	34	430	450
Surface Water	1.4	1.4	0.16	0.17

[a] These values represent the weighted average fish tissue concentration by the percent consumption of trophic levels 3 and 4.

The pasture farm scenario outputs concentrations over time for milk and beef, eggs and chicken or poultry, fish, surface water, soil, and groundwater. Groundwater concentrations range from 0.026 to 4.3 ng/L for PFOA and effectively 0 to 2 ng/L for PFOS. Surface water concentrations range from 0.86 to 10 ng/L for PFOA and 0.16 to 8.5 ng/L for PFOS. Soil concentrations range from 3 to 790 ng/kg for PFOA and 21 to 1,100 ng/kg for PFOS. Fish tissue concentrations range from 29 to 340 for PFOA and 160 to 8,300 ng/kg for PFOS. Milk concentrations range from 1.2 to 46 ng/L for PFOA and 1.9 to 41 ng/L for PFOS. Beef concentrations range from 0.7 to 32 ng/kg for PFOA and 11 to 290 ng/kg for PFOS. Egg concentrations range from 2.9 to 230 ng/kg for PFOA and 13 to 570 ng/kg for PFOS. Finally, chicken ranges from 0.67 to 5.4 ng/kg for PFOA and 1.4 to 60 ng/kg for PFOS.

The trends in soil and groundwater concentrations for PFOA and PFOS seen in the pasture farm model are similar to those seen in the crop farm model, where PFOA concentrations are higher in groundwater and PFOS concentrations are higher in soils; however, maximum estimated soil concentrations are higher in pasture than crop farms. The pasture model does not include tilling of biosolids into the top 20 cm of soil, which results in slightly lower groundwater concentrations and higher soil concentrations. The higher soil concentrations result in a higher loading of runoff into surface water, which results in higher fish tissue concentrations for PFOS. All modeled groundwater results for PFOS would fall below the current MRL for EPA drinking water methods, but the low K_{oc} PFOA results for some climate scenarios would exceed the existing MRLs.

Trends in surface water and fish concentrations for PFOA and PFOS are also similar between the crop farm model and the pasture farm model. In general, the lack of tilling in the pasture model results in more PFOA and PFOS at the surface, available for erosion and runoff into the nearby waterbody. This correspondingly allows for more PFOA and PFOS to be available for fish uptake.

Dairy cows can be exposed to PFOA and PFOS through their feed, forage materials, drinking water, and soil exposure. This model uses uptake factors for lactating dairy cows when calculating both meat and milk concentrations, and assumes that cows are eating non-contaminated grain, but contaminated silage, forage (grass), water, and soil. Overall, high K_{oc} settings result in higher PFOA and PFOS milk and beef concentrations than low K_{oc} settings. Higher K_{oc} settings result in more PFOA and PFOS partitioning to the soils, which in this model also allows more PFOA and PFOS to be available for uptake into forage and silage. Compared to feed, soil is a less significant vector of exposure to cows. A 2012 FDA survey of PFAS concentrations in commercially available milk used a method with MDLs of 120 ppt for PFOA and 130 ppt for PFOS (FDA, 2012); all modeled concentrations fall below these detection thresholds. A more recent dataset from the FDA total diet study (released in 2023) had MDLs of 24 ng/kg for PFOA and 28 ng/kg for PFOS, which was applicable for beef samples (FDA 2023). The modeled results for PFOS were consistently above that MDL, but results for PFOA were often below the MDL. It is important to note that the beef results for PFOA are modeling uptake from dairy cows into muscle; a different BTF would be needed to understand PFOA accumulation into the edible tissues of cows typically raised for beef. Additional data on PFOA and PFOS uptake into beef would help to reduce the uncertainty around these modeled results.

Chickens can also be exposed to PFOA and PFOS through their feed, forage materials, drinking water, and soil exposure. This model uses uptake factors for laying hens when calculating both the egg and meat concentrations. Similar to the cow results, chicken results show that there are higher modeled egg and meat concentrations when K_{oc} is high and in dry climate conditions, where more PFOA and PFOS are retained in the soil. Again, a recent FDA total diet study (FDA 2023) had MDLs of 24 ng/kg for PFOA and 28 ng/kg for PFOS, which was applicable for egg samples. Modeled egg concentrations for PFOS are consistently above that MRL, but modeled egg concentrations for PFOA are sometimes below that MRL.

For reference, the exposures for each pathway for the pasture farm are presented In Tables 24 and 25 in units of ng/kg-day. These exposures are calculated using the consumption rates described in Section 2.9.3.8 as well as other factors described in Sections 2.9.3.9 through 2.9.3.12. The LADD is used for calculating cancer risk and the ADD for noncancer hazard.

Table 24. PFOA Exposures for Pasture Farm (ng/kg-day): LADD and ADD

	Low Koc				High Koc			
	Adult		Child		Adult		Child	
Pathway	LADD	ADD	LADD	ADD	LADD	ADD	LADD	ADD
Dry Climate								
Beef	0.0011	0.012	0.0015	0.016	0.0067	0.051	0.0088	0.067
Eggs	0.0026	0.029	0.0026	0.029	0.022	0.16	0.022	0.16
Fish[a]	0.022	0.16	0.025	0.19	0.0091	0.067	0.011	0.079
Groundwater	0.0052	0.038	0.0054	0.04	4.7E-05	0.00035	5.0E-05	0.00036

Pathway	Low Koc				High Koc			
	Adult		Child		Adult		Child	
	LADD	ADD	LADD	ADD	LADD	ADD	LADD	ADD
Milk	0.014	0.15	0.025	0.27	0.073	0.55	0.13	1.0
Chicken	9.6E-05	0.0011	0.00017	0.0019	0.00078	0.006	0.0014	0.011
Soil	1.0E-06	1.3E-05	1.6E-05	0.00019	1.3E-05	9.9E-05	0.0002	0.0015
Surface water	0.018	0.14	0.019	0.14	0.0077	0.057	0.008	0.06
Moderate Climate								
Beef	0.00022	0.0021	0.00029	0.0028	0.00093	0.0091	0.0012	0.012
Eggs	0.00041	0.0045	0.00041	0.0045	0.0029	0.029	0.0029	0.029
Fish[a]	0.0038	0.030	0.0045	0.035	0.0032	0.024	0.0037	0.028
Groundwater	0.0078	0.057	0.0082	0.06	0.00049	0.0036	0.00051	0.0037
Milk	0.0028	0.026	0.0051	0.047	0.010	0.10	0.019	0.18
Chicken	1.5E-05	0.00016	2.7E-05	0.0003	0.00011	0.0011	0.00019	0.0019
Soil	8.3E-08	1.5E-06	1.3E-06	2.4E-05	1.7E-06	1.7E-05	2.6E-05	0.00027
Surface water	0.0033	0.026	0.0034	0.027	0.0027	0.021	0.0028	0.021
Wet Climate								
Beef	0.00015	0.0014	0.00020	0.0018	0.00064	0.0067	0.00084	0.0088
Eggs	0.00028	0.0029	0.00028	0.0029	0.0019	0.021	0.0019	0.021
Fish[a]	0.0023	0.018	0.0027	0.022	0.0019	0.014	0.0022	0.016
Groundwater	0.0047	0.035	0.0049	0.036	0.0014	0.010	0.0015	0.011
Milk	0.0019	0.017	0.0036	0.031	0.0071	0.074	0.013	0.13
Chicken	1.0E-05	0.00011	1.8E-05	0.0002	7.1E-05	0.00076	0.00013	0.0014
Soil	5.1E-08	9.7E-07	7.8E-07	1.5E-05	1.1E-06	1.2E-05	1.7E-05	0.00018
Surface water	0.0020	0.016	0.0021	0.016	0.0016	0.012	0.0016	0.012

Table 25. PFOS Exposures for Pasture Farm (ng/kg-day): LADD and ADD

Pathway	Low Koc				High Koc			
	Adult		Child		Adult		Child	
	LADD	ADD	LADD	ADD	LADD	ADD	LADD	ADD
Dry Climate								
Beef	0.027	0.23	0.035	0.30	0.062	0.47	0.081	0.61
Eggs	0.016	0.14	0.016	0.14	0.053	0.40	0.053	0.40
Fish[a]	0.52	3.9	0.61	4.5	0.016	0.11	0.018	0.13
Groundwater	0.00041	0.0030	0.00043	0.0031	1.1E-33	8.2E-33	1.2E-33	8.6E-33
Milk	0.032	0.29	0.059	0.53	0.065	0.50	0.12	0.91
Chicken	0.0026	0.023	0.0047	0.041	0.0087	0.066	0.016	0.12
Soil	4.9E-06	4.3E-05	7.4E-05	0.00066	1.9E-05	0.00014	0.00029	0.0022
Surface water	0.015	0.11	0.016	0.12	0.00043	0.0032	0.00045	0.0033
Moderate Climate								
Beef	0.0063	0.052	0.0083	0.069	0.038	0.29	0.050	0.38
Eggs	0.0023	0.023	0.0023	0.023	0.032	0.25	0.032	0.25
Fish[a]	0.15	1.2	0.18	1.4	0.015	0.11	0.017	0.13
Groundwater	0.0021	0.015	0.0022	0.016	1.2E-06	9.1E-06	1.3E-06	9.5E-06
Milk	0.0085	0.069	0.015	0.13	0.041	0.31	0.075	0.57
Chicken	0.00039	0.0038	0.0007	0.0068	0.0053	0.041	0.0096	0.074
Soil	5.1E-07	5.7E-06	7.7E-06	8.7E-05	1.1E-05	8.9E-05	0.00017	0.0013
Surface water	0.0044	0.034	0.0046	0.035	0.00041	0.0030	0.00043	0.0032
Wet Climate								
Beef	0.0024	0.034	0.0032	0.044	0.025	0.19	0.033	0.25
Eggs	0.0012	0.015	0.0012	0.015	0.021	0.16	0.021	0.16
Fish[a]	0.087	0.66	0.10	0.77	0.011	0.081	0.012	0.095
Groundwater	0.0036	0.026	0.0038	0.027	2.1E-05	0.00016	2.2E-05	0.00016
Milk	0.0031	0.044	0.0057	0.080	0.027	0.20	0.049	0.37
Chicken	0.00020	0.0025	0.00037	0.0045	0.0034	0.026	0.0062	0.048
Soil	3.5E-07	4.3E-06	5.4E-06	6.5E-05	7.4E-06	5.7E-05	0.00011	0.00086
Surface water	0.0025	0.019	0.0026	0.020	0.00029	0.0022	0.00030	0.0023

3.1.3 Reclamation Site

The reclamation site model is similar to the pasture farm model, except the reclamation site models a single large application of biosolids rather than ongoing applications of biosolids at an agronomic rate. Assumptions about the duration of time a family spends living near the reclamation site are the same as described for the crop and pasture farm models (10 years). The reclamation site model is also run in dry, moderate, and wet climate settings. This modeling exercise assumes that a dairy farm is established at the site, which is thought to represent a location being reclaimed from over-grazing. However, any of the pathways related to soil, surface water, groundwater, and fish are relevant to many other reclamation scenarios of a similar size (one application of biosolids to 80 acres of remediated land).

Table 26. PFOA Media Concentrations for Reclamation Site (ppt): Maximum 10- and 1-year Averages

Pathway	Low Koc		High Koc	
	10-yr	1-yr	10-yr	1-yr
Dry Climate				
Beef	3.8	7.3	7.0	7.6
Eggs	26	56	52	58
Fish[a]	57	58	18	18
Groundwater	0.17	0.17	0.003	0.003
Milk	5.7	11	10	11
Chicken	0.61	1.3	1.2	1.3
Soil	84	200	180	200
Surface water	1.7	1.7	0.55	0.55
Moderate Climate				
Beef	0.8	4.7	5.7	7.5
Eggs	5.5	36	40	56
Fish[a]	6.5	8.8	14	15
Groundwater	0.054	0.42	0.023	0.023
Milk	1.2	6.8	8.3	11
Chicken	0.13	0.83	0.93	1.3
Soil	18	120	130	190
Surface water	0.19	0.26	0.42	0.45
Wet Climate				
Beef	0.35	2.1	4.6	6.7
Eggs	2	15	31	49
Fish[a]	8.4	15	14	16
Groundwater	0.24	2.4	0.032	0.032
Milk	0.54	3	6.7	9.7
Chicken	0.046	0.34	0.72	1.1
Soil	5.1	48	97	160
Surface water	0.25	0.44	0.42	0.48

[a] These values represent the weighted average fish tissue concentration by the percent consumption of trophic levels 3 and 4.

Table 27. PFOS Media Concentrations for Reclamation Site (ppt): Maximum 10- and 1-year Averages

Pathway	Low Koc		High Koc	
	10-yr	1-yr	10-yr	1-yr
Dry Climate				
Beef	45	51	48	51
Eggs	83	100	92	100
Fish[a]	1200	1300	33	33
Groundwater	0.032	0.032	2.1E-32	2.1E-32
Milk	6.6	7.2	6.8	7.2
Chicken	8.7	10	9.6	10
Soil	160	200	180	200
Surface water	1.3	1.3	0.032	0.032

Pathway	Low Koc		High Koc	
	10-yr	1-yr	10-yr	1-yr
Moderate Climate				
Beef	22	48	42	50
Eggs	34	89	77	98
Fish[a]	480	580	30	31
Groundwater	0.13	0.13	1.1E-5	1.1E-5
Milk	3.4	6.9	6	7.1
Chicken	3.5	9.4	8.1	10
Soil	62	180	150	200
Surface water	0.49	0.59	0.03	0.03
Wet Climate				
Beef	20	40	36	46
Eggs	26	71	62	88
Fish[a]	660	850	28	30
Groundwater	0.12	0.12	3.1E-5	3.1E-5
Milk	3.2	6.1	5.2	6.6
Chicken	2.8	7.4	6.5	9.2
Soil	45	140	120	180
Surface water	0.68	0.87	0.027	0.029

[a] These values represent the weighted average fish tissue concentration by the percent consumption of trophic levels 3 and 4.

The reclamation scenario outputs concentrations over time for milk and beef, eggs and chicken or poultry, fish, surface water, soil, and groundwater. Groundwater concentrations range from 0.003 to 2.4 ng/L for PFOA and effectively 0 to 0.13 ng/L for PFOS. Surface water concentrations range from 0.19 to 1.7 ng/L for PFOA and 0.027 to 1.3 ng/L for PFOS. Fish tissue concentrations range from 6.5 to 58 for PFOA and 28 to 1300 ng/kg for PFOS. Soil concentrations range from 5.1 to 200 ng/kg for PFOA and 45 to 200 ng/kg for PFOS. Milk concentrations range from 0.54 to 11 ng/L for PFOA and 3.2 to 7.2 for PFOS. Beef concentrations range from 0.35 to 7.6 ng/kg for PFOA and 20 to 51 ng/kg for PFOS. Egg concentrations range from 2 to 58 ng/kg for PFOA and 26 to 100 ng/kg for PFOS. Finally, chicken ranges from 0.046 to 1.3 ng/kg for PFOA and 2.8 to 10 ng/kg for PFOS.

Groundwater concentrations in the remediation scenario are lower than those modeled in the pasture farm model. Though the remediation scenario assumed a higher rate of biosolids application than the pasture farm scenario (50 vs 10 DMT per field hectare), the remediation scenario only included a single application of biosolids, while the pasture farm scenario included annual applications for 40 years. This modeling suggests that a single application of low concentration biosolids is unlikely to result in a detectable PFOA concentration in groundwater, though this outcome is more likely in scenarios where the underlying soils had a low K_{oc} (meaning low soil sorption), such as sandy soils or soils damaged by human activity in a way that results in geochemical conditions less conducive to soil sorption. One example of a soil condition resulting in low PFOA and PFOS sorption is high soil pH; at normal soil pH ranges, the pKa of PFOA and PFOS indicate they would be negatively charged such that lower soil pH results in higher rates of nonspecific anion absorption (Oliver et al., 2019). Given that soil remediation can occur in a variety of depleted or disturbed sites, it is possible that these low sorption conditions are relevant to many biosolids reuse scenarios where the biosolids are used to remediate disturbed soils.

Soil, surface water, and fish tissue concentrations are also lower in the remediation scenario than in the pasture farm scenario. However, there are smaller differences between the location settings and high K_{oc} and low K_{oc} settings in the remediation scenario than in the pasture farm scenario. This is likely because, when biosolids are only applied one time, the maximum PFOA and PFOS concentrations are reached more rapidly, and differences in leaching potential from the soil over time are less impactful on the maximum observed concentration.

Beef, chicken, eggs, and milk concentrations are also lower in the remediation scenario than the pasture farm scenario, where PFOA and PFOS concentration have time over repeated applications to accumulate in soils. Because the media concentrations in these scenarios are lower than in the pasture farm setting, many (but not all) of the modeled concentrations would fall below currently available MDLs. However, given the high bioaccumulation of PFOS in fish and eggs, these media would consistently have detectable concentrations of PFOS in this scenario.

For reference, the exposures for each pathway for the reclamation site are presented In Tables 28 and 29 in units of ng/kg-day. These exposures are calculated using the consumption rates described in Section 2.9.3.8 as well as other factors described in Sections 2.9.3.9 through 2.9.3.12. The LADD is used for calculating cancer risk and the ADD for noncancer hazard.

Table 28. PFOA Exposures for Reclamation Site (ng/kg-day): LADD and ADD

	Low Koc				High Koc			
	Adult		Child		Adult		Child	
Pathway	LADD	ADD	LADD	ADD	LADD	ADD	LADD	ADD
Dry Climate								
Beef	0.00084	0.012	0.0011	0.015	0.0015	0.012	0.0020	0.016
Eggs	0.0025	0.039	0.0025	0.039	0.0050	0.040	0.0050	0.040
Fish[a]	0.0037	0.027	0.0043	0.032	0.0012	0.0087	0.0014	0.010
Groundwater	0.00032	0.0023	0.00033	0.0024	5.4E-06	4.0E-05	5.7E-06	4.1E-05
Milk	0.0094	0.13	0.017	0.23	0.017	0.13	0.031	0.24
Chicken	9.3E-05	0.0014	0.00017	0.0026	0.00018	0.0015	0.00033	0.0027
Soil	1.4E-06	2.5E-05	2.2E-05	0.00038	3.1E-06	2.5E-05	4.7E-05	0.00038
Surface water	0.0031	0.023	0.0033	0.024	0.0010	0.0074	0.0010	0.0077
Moderate Climate								
Beef	0.00017	0.0076	0.00023	0.0099	0.0012	0.012	0.0016	0.016
Eggs	0.00053	0.025	0.00053	0.025	0.0038	0.039	0.0038	0.039
Fish[a]	0.00042	0.0041	0.00049	0.0048	0.00091	0.0072	0.0011	0.0084
Groundwater	0.00010	0.0056	0.00010	0.0059	4.1E-05	0.00030	4.3E-05	0.00032
Milk	0.0019	0.082	0.0035	0.15	0.014	0.13	0.025	0.24
Chicken	1.9E-05	0.00091	3.5E-05	0.0017	0.00014	0.0014	0.00025	0.0026
Soil	3.0E-07	1.5E-05	4.6E-06	0.00023	2.2E-06	2.4E-05	3.4E-05	0.00037
Surface water	0.00035	0.0035	0.00037	0.0037	0.00078	0.0061	0.00081	0.0064
Wet Climate								
Beef	7.6E-05	0.0033	0.00010	0.0043	0.0010	0.011	0.0013	0.014
Eggs	0.00019	0.010	0.00019	0.010	0.0030	0.034	0.0030	0.034
Fish[a]	0.00054	0.0070	0.00063	0.0081	0.00091	0.0076	0.0011	0.0089
Groundwater	0.00045	0.032	0.00047	0.034	5.8E-05	0.00042	6.1E-05	0.00044
Milk	0.00088	0.036	0.0016	0.066	0.011	0.12	0.02	0.21
Chicken	7.0E-06	0.00037	1.3E-05	0.00068	0.00011	0.0012	0.0002	0.0023
Soil	8.8E-08	6.0E-06	1.3E-06	9.2E-05	1.7E-06	2.0E-05	2.5E-05	0.00031
Surface water	0.00046	0.0059	0.00048	0.0062	0.00077	0.0065	0.00080	0.0068

Table 29. PFOS Exposures for Reclamation Site (ng/kg-day): LADD and ADD

	Low Koc				High Koc			
	Adult		Child		Adult		Child	
Pathway	LADD	ADD	LADD	ADD	LADD	ADD	LADD	ADD
Dry Climate								
Beef	0.0099	0.082	0.013	0.11	0.01	0.082	0.014	0.11
Eggs	0.0079	0.070	0.0079	0.07	0.0088	0.070	0.0088	0.070
Fish[a]	0.080	0.59	0.093	0.69	0.0021	0.016	0.0025	0.018
Groundwater	5.8E-05	0.00042	6.1E-05	0.00044	3.9E-35	2.8E-34	4.0E-35	2.9E-34
Milk	0.011	0.087	0.020	0.16	0.011	0.087	0.020	0.16
Chicken	0.0013	0.011	0.0024	0.021	0.0015	0.012	0.0026	0.021

	Low Koc				High Koc			
	Adult		Child		Adult		Child	
Pathway	**LADD**	**ADD**	**LADD**	**ADD**	**LADD**	**ADD**	**LADD**	**ADD**
Soil	2.8E-06	2.5E-5	4.2E-05	0.00038	3.1E-06	2.5E-05	4.8E-05	0.00038
Surface water	0.0023	0.017	0.0024	0.018	5.8E-05	0.00044	6.1E-05	0.00046
Moderate Climate								
Beef	0.0048	0.076	0.0063	0.10	0.0092	0.081	0.012	0.11
Eggs	0.0032	0.063	0.0032	0.063	0.0074	0.069	0.0074	0.069
Fish[a]	0.031	0.27	0.036	0.32	0.0020	0.015	0.0023	0.017
Groundwater	0.00024	0.0018	0.00025	0.0019	2.0E-08	1.5E-07	2.1E-08	1.5E-07
Milk	0.0056	0.082	0.010	0.15	0.0099	0.085	0.018	0.16
Chicken	0.00053	0.010	0.00097	0.019	0.0012	0.011	0.0022	0.021
Soil	1.1E-06	2.2E-05	1.6E-05	0.00034	2.6E-06	2.5E-05	4.0E-05	0.00037
Surface water	0.00090	0.0079	0.00094	0.0082	5.4E-05	0.00041	5.7E-05	0.00043
Wet Climate								
Beef	0.0043	0.065	0.0056	0.085	0.0078	0.074	0.010	0.098
Eggs	0.0025	0.049	0.0025	0.049	0.0059	0.062	0.0059	0.062
Fish[a]	0.043	0.40	0.050	0.47	0.0018	0.014	0.0021	0.017
Groundwater	0.00021	0.0015	0.00022	0.0016	5.6E-08	4.1E-07	5.9E-08	4.3E-07
Milk	0.0052	0.073	0.0095	0.13	0.0086	0.079	0.016	0.15
Chicken	0.00041	0.0081	0.00075	0.015	0.00098	0.010	0.0018	0.018
Soil	7.7E-07	1.8E-05	1.2E-05	0.00027	2.0E-06	2.2E-05	3.1E-05	0.00033
Surface water	0.0012	0.012	0.0013	0.012	5.0E-05	0.00039	5.3E-05	0.00041

3.1.4 Sewage Sludge Disposal Site

The sewage sludge disposal site models the fate and transport of PFOA and PFOS after they are disposed of in an unlined, lined with a composite liner, or clay-lined impoundment. This scenario assumes that the biosolids being disposed of are not dewatered because this a common practice across the U.S. and the practice more likely to result in groundwater infiltration risks. The model considers infiltration from the impoundment through soil and into groundwater. The model then calculates PFOA and PFOS concentrations in groundwater that is used for drinking water. The results in the table below report groundwater concentrations in wet, moderate, and dry climates in a well screened up to 2 m below the water table and 5 m distance from the impoundment site. These climate scenarios also represent the varied soil types, depths to groundwater, hydrological conditions that would be expected in these three climate settings. This scenario assumes that an adult's lifetime only includes 10 years living near the impoundment. The following tables show the modeled concentrations of PFOA and PFOS in each disposal site liner type during either a ten year or one year of averaging time. These averaging windows include the maximum concentration year for groundwater.

Table 30. PFOA Groundwater Concentrations for Sludge Disposal Unit (ppt): Maximum 10- and 1-year Averages by Liner Scenario

	Low K_{oc}		High K_{oc}	
Liner	**10-yr**	**1-yr**	**10-yr**	**1-yr**
Dry Climate				
No Liner	25	25	0.075	0.077
Clay liner	21	21	0.049	0.050
Composite liner	0.013	0.014	1.6E-30	1.6E-30
Moderate Climate				
No Liner	8.8	8.9	0.024	0.025
Clay liner	5.8	5.8	0.016	0.016
Composite liner	0.0011	0.0011	1.5E-30	1.5E-30
Wet Climate				
No Liner	16	17	0.17	0.17
Clay liner	12	13	0.077	0.078
Composite liner	0.0041	0.0041	8.7E-13	8.9E-13

Table 31. PFOS Groundwater Concentrations for Sludge Disposal Unit (ppt): Maximum 10- and 1-year Averages by Liner Scenario

	Low K_{oc}		High K_{oc}	
Liner	**10-yr**	**1-yr**	**10-yr**	**1-yr**
Dry Climate				
No Liner	1.3	1.3	0.00046	0.00048
Clay liner	0.91	0.93	0.00031	0.00033
Composite liner	2.3E-06	2.3E-06	2E-32	2E-32
Moderate Climate				
No Liner	0.43	0.44	0.00018	0.00018
Clay liner	0.25	0.25	0.00010	0.00011
Composite liner	4.5E-14	4.6E-14	2.2E-32	2.3E-32
Wet Climate				
No Liner	2.2	2.2	0.0022	0.0023
Clay liner	1.2	1.2	0.00092	0.00097
Composite liner	1.2E-05	1.3E-05	3.2E-32	3.4E-32

The surface disposal scenario outputs groundwater concentrations over time for three types of disposal sites: unlined, clay-lined, and lined with a composite liner. As expected, groundwater concentrations are the highest in unlined surface disposal sites (PFOA from 0.024 to 25 ng/L; PFOS from essentially zero to 2.2 ng/L). Clay-lined surface disposal sites have slightly lower groundwater concentrations than unlined sites. Finally, composite-lined surface disposal sites result in very low groundwater infiltration, with essentially no infiltration of PFOS and only low breakthrough for PFOA (PFOA groundwater concentrations from zero to 0.014 ng/L; PFOS remains essentially zero in all scenarios). Differences in modeled groundwater concentrations between dry, moderate and wet climates reflect the differences in depth to the water table, infiltration rate, and the amount of dilution of the disposal site material with rainfall in each hypothetical setting.

For reference, the exposures for groundwater for the sludge disposal unit are presented In Tables 32 and 33 in units of ng/kg-day. These exposures are calculated using the consumption rates described in Section 2.9.3.8 as well as other factors described in Sections 2.9.3.9 through 2.9.3.12. The LADD is used for calculating cancer risk and the ADD for noncancer hazard.

Table 32. PFOA Exposures for Surface Disposal Site (ng/kg-day): LADD and ADD

	Low Koc				High Koc			
	Adult		Child		Adult		Child	
Liner	LADD	ADD	LADD	ADD	LADD	ADD	LADD	ADD
Dry Climate								
No liner	2.5E-05	0.00018	2.6E-05	0.00019	2.9E-33	2.2E-32	3E-33	2.3E-32
Clay liner	0.046	0.34	0.048	0.35	0.00014	0.001	0.00014	0.0011
Composite liner	0.038	0.28	0.039	0.29	9.1E-05	0.00068	9.5E-05	0.00071
Moderate Climate								
No liner	2.0E-06	1.5E-05	2.1E-06	1.6E-05	2.7E-33	2.1E-32	2.9E-33	2.2E-32
Clay liner	0.016	0.12	0.017	0.12	4.4E-05	0.00033	4.6E-05	0.00034
Composite liner	0.011	0.078	0.011	0.081	2.9E-05	0.00021	3.0E-05	0.00022
Wet Climate								
No liner	7.5E-06	5.5E-05	7.8E-06	5.7E-05	1.6E-15	1.2E-14	1.7E-15	1.2E-14
Clay liner	0.029	0.22	0.03	0.23	0.00031	0.0023	0.00032	0.0024
Composite liner	0.023	0.17	0.024	0.18	0.00014	0.0011	0.00015	0.0011

Table 33. PFOS Exposures for Surface Disposal Site (ng/kg-day): LADD and ADD

	Low Koc				High Koc			
	Adult		Child		Adult		Child	
Liner	LADD	ADD	LADD	ADD	LADD	ADD	LADD	ADD
Dry Climate								
No liner	0.0024	0.018	0.0025	0.019	8.4E-07	6.4E-06	8.8E-07	6.7E-06
Clay liner	0.0017	0.012	0.0018	0.013	5.7E-07	4.4E-06	6E-07	4.6E-06
Composite liner	4.1E-09	3.1E-08	4.3E-09	3.2E-08	3.6E-35	2.7E-34	3.8E-35	2.9E-34
Moderate Climate								
No liner	0.00079	0.0059	0.00083	0.0062	3.2E-07	2.5E-06	3.4E-07	2.6E-06
Clay liner	0.00046	0.0034	0.00048	0.0035	1.8E-07	1.4E-06	1.9E-07	1.5E-06
Composite liner	8.3E-17	6.2E-16	8.6E-17	6.5E-16	4E-35	3.1E-34	4.2E-35	3.2E-34
Wet Climate								
No liner	0.004	0.029	0.0041	0.031	4.0E-06	3.1E-05	4.1E-06	3.2E-05
Clay liner	0.0021	0.016	0.0022	0.016	1.7E-06	1.3E-05	1.8E-06	1.4E-05
Composite liner	2.3E-08	1.7E-07	2.4E-08	1.8E-07	5.8E-35	4.5E-34	6.1E-35	4.7E-34

3.1.5 Implications for Home Gardening

This assessment does not explicitly model how use of Class A_{EQ} biosolids in home gardens could impact soil, fruit and vegetable, and groundwater concentrations. Class A_{EQ} biosolids have no application requirements; they do not need to be applied at agronomic rates. Sizes of home gardens vary greatly but are generally much smaller than a field used for growing crops at even a small commercial farm. The smaller application areas for Class A_{EQ} biosolids at a given site likely reduces concerns over PFOA and PFOS impacts to surface water and groundwater, though if larger amounts of biosolids were bulk applied to a hobby farm or community garden as fertilizer, there could be potential impacts.

There is a high degree of uncertainty in the rates of PFOA and PFOS uptake to fruits and vegetables. With the limited data available, it appears that vegetables like spinach and lettuce are the most likely to uptake PFOA and PFOS, with PFOA exhibiting higher rates of uptake than PFOS. It is conceivable that a home gardener using biosolids-based products in their raised beds or backyard garden could apply enough biosolids, potentially over multiple years, to sufficiently elevate PFOA and PFOS concentrations in soils such that detectable levels of PFOA and PFOS could be found in some fruits and vegetables. It is also possible that a home gardener with backyard chickens could have enough PFOA and PFOS in vegetable scraps, soil, grubs, and grass to result in measurable concentration of PFOA and PFOS in eggs.

Finally, it is possible that homes have been developed on land that was previously used as farmland and that had historic biosolids land application. Homeowners living in these developments could start a garden with or without adding any new biosolids-based products to their garden beds. Given the long residency times for PFOA and PFOS in soils in the crop and pasture modeling scenarios, it is possible that a home gardener could be exposed to PFOA and PFOS in homegrown food or home raised eggs if they are living on land that previously accepted agronomic land application, even if that homeowner does not add any new PFOA or PFOS to their yard.

3.1.6 Other Land Application Use Scenarios

As described in the conceptual model section of this report (Section 2.8), there are numerous potential land application scenarios that have not been explicitly modeled in this report, including biosolids applications to forests, tree farms, road construction sites, golf courses, and more. If these locations are considered “low public contact,” potential pathways of exposure include groundwater used for drinking water, surface water used for drinking water, and fish consumption from an impacted waterbody. If biosolids are applied in an area with potential for soil exposure, this pathway could be relevant as well.

The scenarios modeled in this draft risk assessment are also not designed to explicitly account for exposures that may occur where Class A_{EQ} biosolids are applied at non-agricultural sites. Soil concentrations at sites where Class A_{EQ} biosolids application have occurred may be roughly described by the pasture farming scenario, however, farmers are required to limit application rates for Class A and B biosolids to the nutrient needs for the crop at the farm. Class A_{EQ} biosolids are sold to the general public and landscapers and may be used without an understanding of matching the nutrient need of the soil to the application rate, so over-application is possible. Therefore, it may be appropriate to consider that soil concentrations could more rapidly rise in a Class A_{eq} application scenario than in a crop or pasture farm scenario. This assessment is using a central tendency incidental soil ingestion rate (40 mg/day for children aged 1-5), when incidental soil ingestion is evaluated for children. When creating CERCLA screening values for residential areas an upper percentile rate is used for children (200 mg/day). Conservatism in exposure assessment for Class A_{EQ} biosolids is warranted for children’s incidental soil ingestion exposures given that larger number of children may be exposed at homes, playgrounds, parks or other areas where Class A_{EQ} biosolids may be used in larger proportion than other land application sites like farms.

The trends observed in the modeling performed for remediation sites and farms can inform the types of concentrations expected in some other types of land application scenarios, acknowledging that each land application scenario is unique. For example, annual application of biosolids to a golf course or turf farm, applied at agronomic rates for fertilizing turf grass, is likely to show similar soil, surface water, fish, and groundwater concentration trends as the pasture farm scenario, with the understanding that the size of the biosolids-applied area will linearly scale with the final modeled media concentrations. Similarly, annual application of biosolids to a forest or tree farm could result in similar media concentration trends as the pasture farm scenario, with the caveat that silviculture or forested areas likely have meaningfully different rates of runoff and erosion than a grass field. Additionally, a forested land application scenario could have some amount of PFOA and PFOS uptake into trees, which could result in less mass available for runoff into a nearby waterbody or infiltration into groundwater.

Using biosolids during road construction is a somewhat common practice. For example, a recent report from the City of Juneau, Alaska, explains that there is a growing market for biosolids use as an erosion control technique for construction projects, including road construction (City and Borough of Juneau Wastewater Utility, 2017). The report explains that biosolids pellets can be used to enhance topsoil, to fill void spaces and limit channelized flow of water on roadsides, provide a more permeable surface to promote infiltration, and aid in revegetation along the road. The report adds that dried and pelletized biosolids could be used in a filter sock to prevent water from reaching storm drains as a replacement for a silt fence or straw bale barrier for stormwater control. The modeling included in this report would not capture the fate and transport of PFOA and PFOS when biosolids are used in this manner, though it is possible that soils, surface water, groundwater, and fish may all be impacted from PFOA and PFOS in these settings.

Mine reclamation is another type of biosolids land application that is not explicitly modeled in this assessment. When biosolids are used in mine reclamation, there is generally one or a small number of larger application of biosolids to increase the organic material and/or pH at the site. Former mining sites can vary greatly in size and hydrogeological settings. They can also have more extreme geochemical conditions and soil properties, including very low organic content and potentially high concentrations of metals. These factors would need to be modeled with site-specific information to understand how they are likely to impact the fate and transport of PFOA and PFOS at the site.

3.1.7 Incineration

Current SSIs may not operate at temperatures that are sufficient to completely destroy PFAS compounds to mineralized compounds (CO_2, HF, F_2). Therefore, incineration could result in PFOA or PFOS emissions via either incomplete combustion of those chemicals in the sewage sludge or if other PFAS are only partially destroyed and create PFOA and PFOS or their precursors. Given that SSIs can destroy some proportion of PFOA and PFOS (Winchell et al., 2024), deposition of PFOA and PFOS from an SSI to nearby soils would lead to lower exposures than the land application of equivalently contaminated sewage sludge. However, past sewage sludge assessments (US EPA, 1992) have separately assessed incineration. This assessment is not attempting to create an incineration exposure estimate given the active research and investigation of PFAS destruction efficiency during incineration and potential exposure to PICs.

3.2 Modeled Media Concentrations over Time

The fate and transport models used in this assessment calculate estimates of media concentrations over time with daily resolution. This allows for understanding how PFOA and PFOS might be transported throughout the modeled environment over time. For illustrative purposes, the following sections describe the changes in PFOA and PFOS concentrations over time in the crop or pasture farm model run at the “moderate” climate setting.

3.2.1 Soil Concentrations over Time

The current modeling effort does not take into account the effects of PFOA and PFOS precursor transforming to PFOA and PFOS in soil over time. Studies of biosolids land-application sites with PFAS contamination indicate that the transformation of precursors acts as a long-term source of PFOA and PFOS, well after land application has ceased (Washington et al., 2010; Yoo et al. 2010). That said, modeled soil concentrations over time are still valuable in understanding how soil concentration change as material is added, eroded, taken up into plants and animals, and leached to groundwater.

There are differences in the modeled concentration trends over time for the low and high K_{oc} settings. The low K_{oc} setting at the “moderate” climate crop farm is depicted in **Figure 9**. In this setting, the PFOA is quickly mobilized from the soil, such that levels do not build up with annual additions of biosolids. Despite these low-sorption soil conditions, these models still indicate that PFOS will persist long enough in soils to accumulate over time during the timeframe of biosolids application. However, PFOS concentrations in the topsoil drop quickly after land-application end, and PFOS concentrations averaged across the soil profile also have a steady declining trend.

As seen in **Figure 10** (the high K_{oc} setting), the soil concentrations increase over time as annual biosolids land applications occur. When applications of biosolids stop after 40 years, the concentrations of PFOA and PFOS in soil begin to decline. The rates of decline are significantly faster for the top layer of soil, while the average soil concentration in the top 20 centimeters declines more slowly. The rate of decline is faster for PFOA than PFOS because PFOA is more mobile and thus more leachable from soils. The variability in concentrations over time reflects ongoing natural mixing of the soil and changes in weather over time. In this high K_{oc} setting, PFOS concentrations remain elevated throughout the model run duration (150 years).

In the high K_{oc} scenario for PFOS, the soil concentrations decline along an expected trend line until year 80, when they dip dramatically and appear to rebound. This trend is a known artifact of the numerical modeling used in 3MRA's Land Application Unit module and does not impact the risk calculations in this assessment. In short, the numerical formulation of the LAU's Generic Soil Column Model (GSCM; US EPA, 1999) solves the three components of the governing transport equation—diffusive transport, advective transport, and contaminant decay—in a layered soil column. The advective process moves mass downward through each layer of the soil column with an effective convection velocity corrected for contaminant partitioning to the water and solid phases; this effective convection velocity is heavily influenced by K_{oc}. The advective component of the transport equation moves contaminant mass down to the next layer (and ultimately, out the bottom of the LAU) at discrete time intervals equal to the time it takes for dissolved contaminants to traverse a layer via convective transport. At large K_{oc} (e.g., the 90th percentile K_{oc} for PFOS), the contaminant's effective velocity is very slow and the amount of mass sorbed to soil is much greater, resulting a relatively large amount of sorbed mass leaving the system at once at discrete time intervals and resulting in the sharp drops at predictable intervals visible in the media concentration charts for PFOS with high K_{oc}. The magnitude and frequency of these oscillations are directly related to the magnitude of the K_{oc}: this numerical artifact is always present, but with smaller K_{oc}, the oscillation is much smaller and more frequent and so not distinguishable from numerical noise. Regardless of the size of the oscillations, they do not affect the risk results, as those are based on the 1-year average concentrations at the peak (for noncancer) or averaged over the 10-year period that is centered on the peak (for cancer). The peak soil concentration is always close to year 40 in the pasture farm and crop farm scenarios, when biosolids stop being added to the field and before the oscillatory behavior becomes apparent.

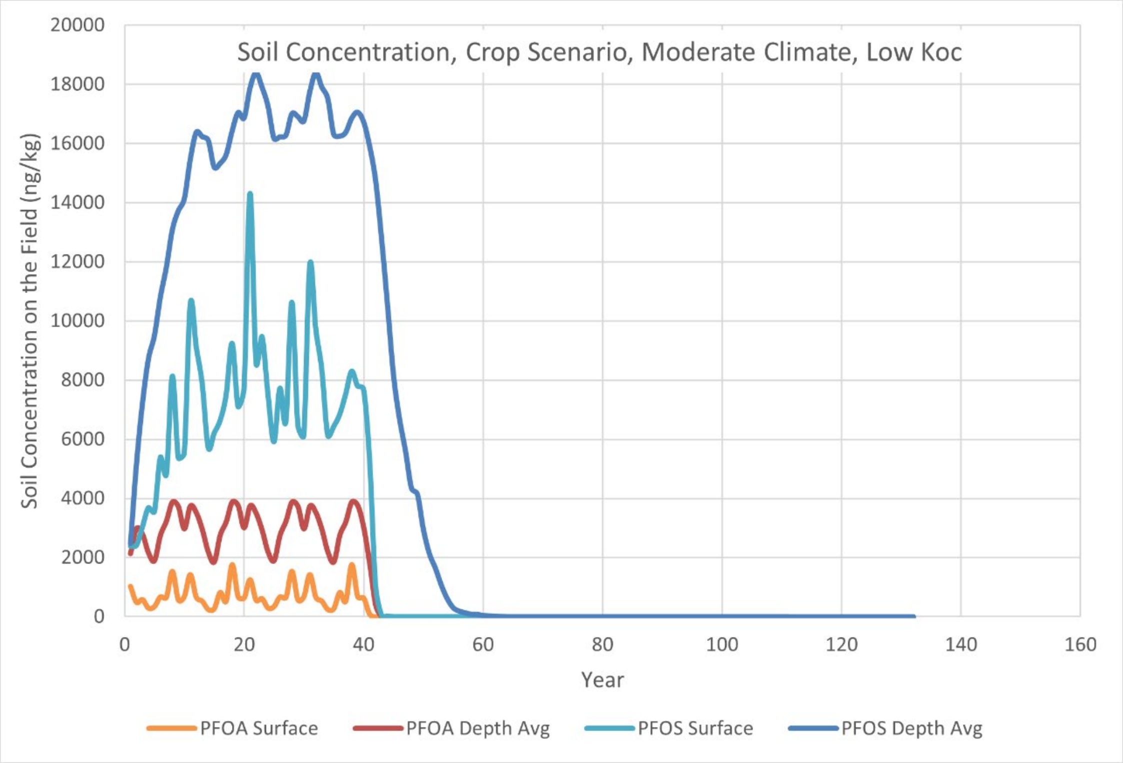

Figure 9. Plot of PFOA and PFOS concentrations over time in the "moderate" climate crop farm scenario with the low K_{oc} setting, assuming biosolids application ceases after 40 years.

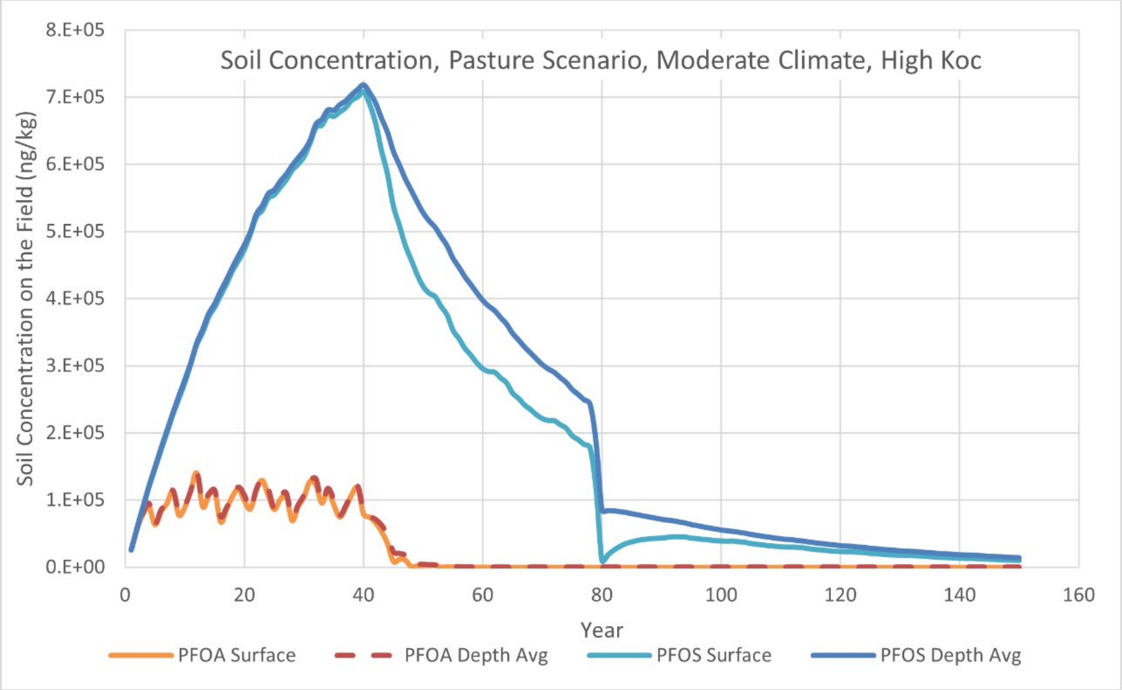

Figure 10. Plot of PFOA and PFOS concentrations in soil over time in the "moderate" climate pasture farm scenario with the high K_{oc} setting, assuming biosolids application ceases after 40 years.

3.2.2 Surface Water Concentrations over Time

The trends in modeled surface water concentrations also change over time depending on if the farm is modeled using the low or high K_{oc} setting. In the low K_{oc} setting (**Figure 11**), PFOA and PFOS concentrations in surface water steadily increase over time up until land application stops after 40 years. PFOA is more leachable into the aqueous phase and has a larger degree of transport to surface water in the dissolved phase; for this reason, concentrations of PFOA in surface water are more responsive to changes in precipitation. After land application ends, PFOA and PFOS concentrations decrease in surface water rapidly over the next 20 to 40 years, and then more slowly from model years 80 to 150.

The PFOA and PFOS surface water trends are different in the high K_{oc} setting (**Figure 12**), where PFOS concentrations rise and fall slowly in surface water compared to PFOA concentrations. This trend likely reflects the fact that the high sorption scenario for PFOS results in more retention in the soil column and less mobilization into surface water.

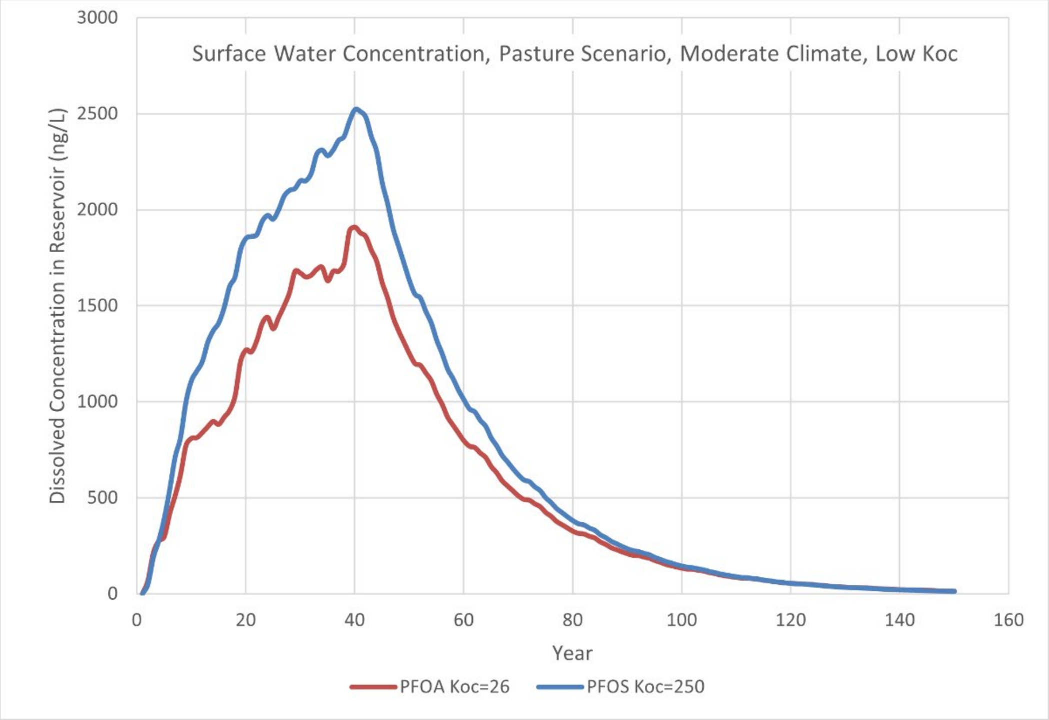

Figure 11. PFOA and PFOS concentrations over time in the low K_{oc}, pasture farm, moderate climate setting.

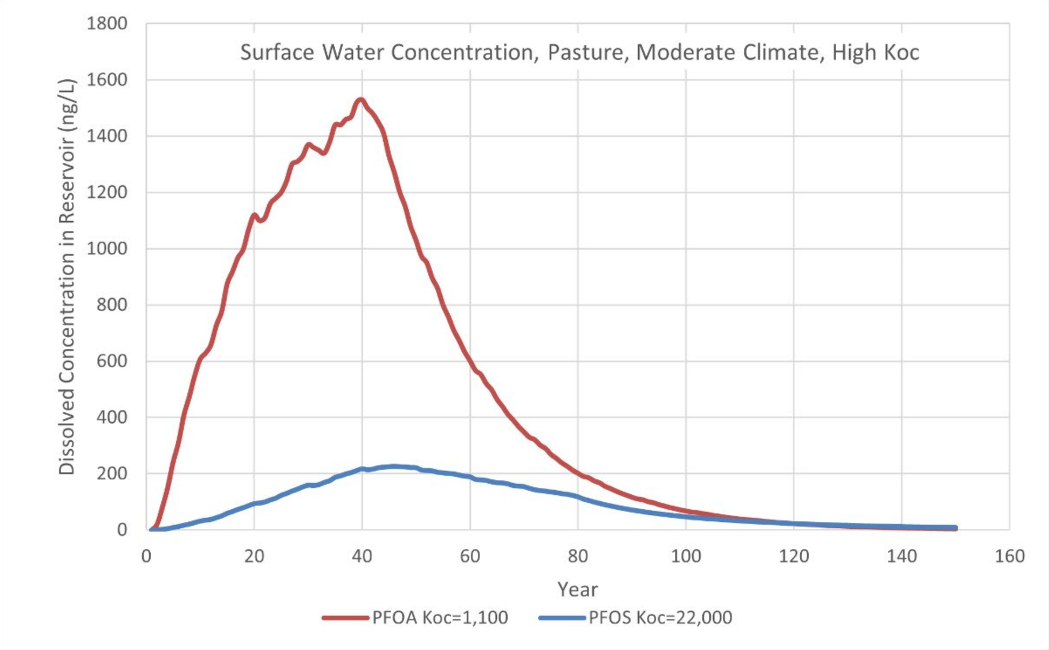

Figure 12. PFOA and PFOS concentrations over time in the high K_{oc}, pasture farm, moderate climate setting.

3.2.3 Groundwater Concentrations over Time

Both the low K_{oc} and high K_{oc} settings for PFOA and PFOS show that it takes a considerable amount of time for these chemicals to move from the soil (where they are applied in biosolids) into the vadose zone, and through to groundwater. In the low K_{oc} setting (**Figure 13**), models indicate it takes between 10 and 30 years for PFOA and 500 and 1000 years for PFOS to reach a hypothetical well five meters away from the field. The well depth was selected after reviewing the concentration profile in groundwater at depth increments of 0.5 meters to 2.0 meters below the water table and selecting this highest concentration depth for this distance from the field. Assessment of the concentration of PFOA and PFOS with depth indicated that the concentration is relatively constant down to 6 to 8 meters below the water table, so the choice of selecting the maximum value in the top 2.0 meters of the aquifer does not significantly impact the assessment (see Appendix C).

In the high K_{oc} setting (**Figure 14**), the models indicate that it takes between 300 to 400 years for PFOA and 6,000 to 8,000 years for PFOS to reach that hypothetical well. Empirical observations of groundwater concentrations in monitoring wells and drinking water wells near biosolids land-application sites indicate that these modeled timeframes for higher K_{oc} settings are likely incorrect (too long) by orders of magnitude (see Section 5.3 and Appendix C for more details).

The leaching potential for PFOA and PFOS at any specific site can be highly variable due to a variety of factors, many of which are not captured in this draft risk assessment. For example, a recent study investigated the effects of microbial weathering on PFAS partitioning over time after biosolids land application to examine the fate and transport of PFAS leaching from biosolids into the environment (Lewis et al., 2023). Results revealed that microbial weathering plays a role in PFAS partitioning, contributing to the biodegradation of organic matter and leading to an increased potential for PFAS leaching to groundwater. The weathering of the biosolids matrix is not taken into account in this assessment's groundwater models. Additionally, another study showed that the dry-wet and freeze-thaw cycles that are a natural occurrence in subsurface soils can lead to increased PFOA leaching

(Borthakur et al., 2021). It is not entirely clear how the freeze-thaw cycles result in more leaching, and there are no models available that incorporate this effect. A third recent study examined how colloidal transport mechanisms may facilitate faster rates of PFAS leaching (Bierbaum et al., 2023). In general, the existence of preferential flow pathways in soils, sometimes called “macropores,” may also facilitate faster leaching than is modeled in this assessment. Colloidal transport mechanisms and preferential flow pathways like cracks, soil type boundaries, or worm and insect tunnels are not accounted for in the groundwater model used in this assessment. EPA will continue evaluating the availability of groundwater and vadose zone models as this assessment is finalized.

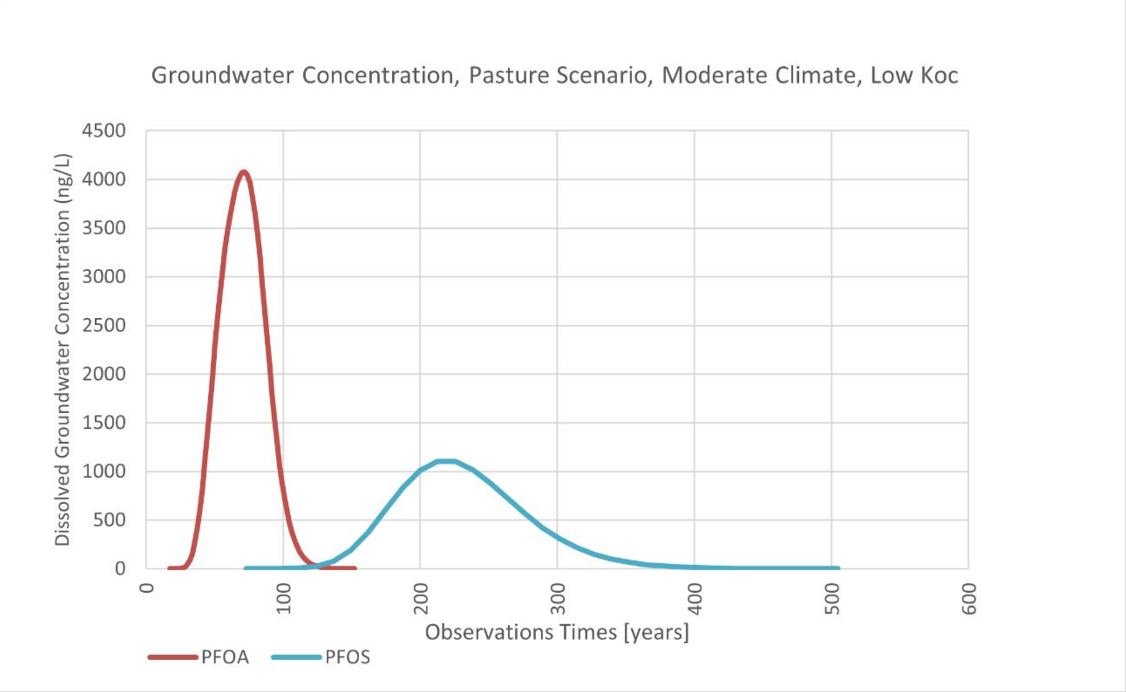

Figure 13. PFOA and PFOS concentrations over time in the low K_{oc}, pasture farm, moderate climate setting.

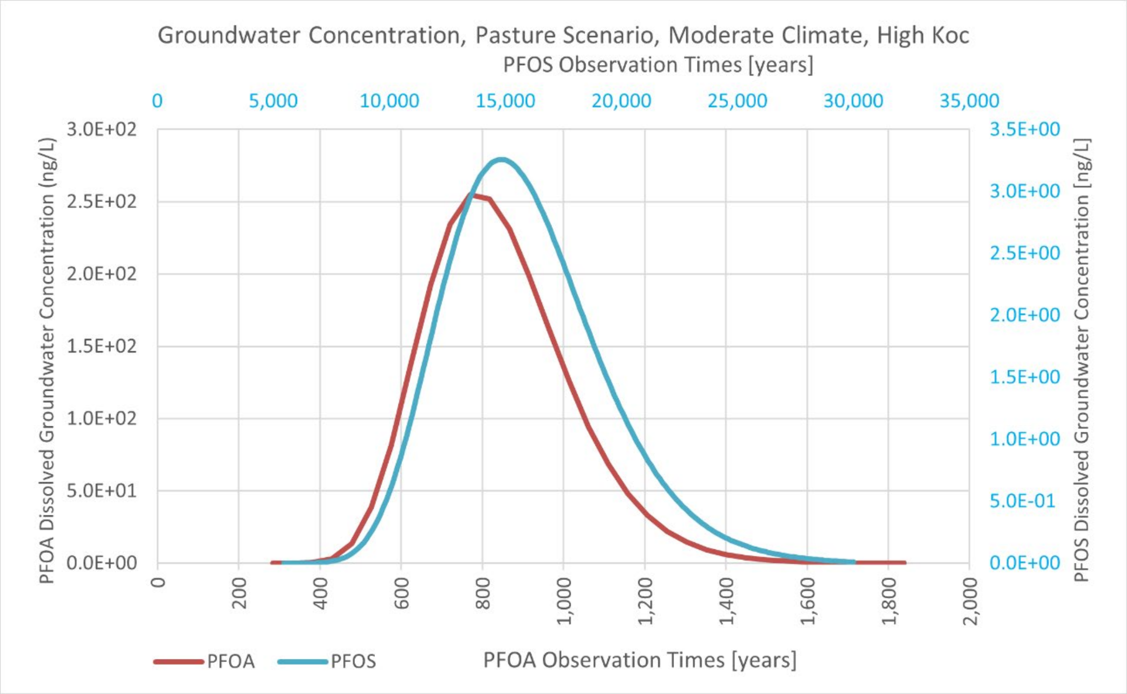

Figure 14. PFOA and PFOS concentrations over time in the high K_{oc}, pasture farm, moderate climate setting.

4 RISK CHARACTERIZATION

The following sections integrate modeled media concentration results with human intake rates for each media and human health effects thresholds to describe risks to receptors. Risks are discussed for each of the biosolids use or disposal scenarios outlined in Section 2.8. As described in Section 5.3, the exposure results from fate and transport modeling are sensitive to the parameters associated with the climate and K_{oc}. For this reason, risk results are presented in the same manner as media concentrations, with results presented for each climate (dry, moderate, and wet) and for a low K_{oc} and high K_{oc}. Risks are further disaggregated into hazard quotients (HQs) for non-cancer effects and cancer risk levels (CRLs).

4.1 Methods for Estimating Human Health Hazard and Cancer Risk

Cancer risk is characterized by calculating the lifetime excess cancer risk for the target population, which is the increased probability that a member of that population will develop cancer over a lifetime because of exposure to the pollutant. To evaluate oral exposures to carcinogens, the LADD is used. The LADD is calculated by finding the modeling year with the highest average daily dose for the given media (*i.e.*, groundwater, surface water, soil), and calculating the average daily dose for the ten years around the maximum concentration modeling year. The result is a lifetime average daily dose that spans a 10-year residency on the site and is centered around the year associated with the highest dose for groundwater, surface water or soil. The models run for 150 years and assume that a lifetime only includes 10 years on the contaminated site (see section 2.9.3.12 for discussion of the duration of exposure modeling), with the remainder of the 60 years taking place in a location with zero PFOA and PFOS exposure. For example, if the highest concentration of PFOA or PFOS in groundwater used as drinking water does not occur until forty years after biosolids application on a farm field begins, the lifetime cancer risk is calculated by averaging the daily dose of exposure from drinking water spanning from model year 35 to model year 44. That daily dose average is then scaled down to 350 days per year (to account for travel time away from the residence) and normalized over a 70-year lifetime to calculate the LADD. This LADD is multiplied by the CSF to calculate the excess lifetime cancer risk from using impacted groundwater as drinking water. This approach is used to calculate LADDs for children and adults, and assumes that there are no exposures to PFOA or PFOS from drinking water when the resident is not living at the contaminated residence or when the resident is traveling away from the home during their period of residence. These assumptions about residency time and off-site exposure could result in an underestimation of risk.

The EPA does not have a single cancer risk level that is used for risk assessments, but generally targets cancer risk levels of 1 in 100,000 (1×10^{-5}) or 1 in 1 million (1×10^{-6}) for carcinogens, depending on the size of the impacted population (*e.g.*, US EPA, 2000a). Given that this central tendency modeling exercise is parameterized with median values and is modeling risks for PFOA and PFOS near the detection limit for biosolids (1 ppb), the EPA anticipates that these model scenarios may be applicable across many biosolids use and disposal sites in the U.S.. Further, because the starting concentration of PFOA and PFOS are linearly related to the modeled media concentrations, a scenario modeled to exceed a 1-in-1-million cancer risk level in this draft risk assessment would exceed a 1-in-100,000 cancer risk level if the starting concentration for PFOA or PFOS were 10 ppb. Monitoring of sewage sludge in states like Michigan indicate that biosolids with either PFOA or PFOS exceeding 10 ppb are common (see Section 2.4). Therefore, this draft risk assessment will highlight excess cancer risks exceeding 1-in-1-million (1×10^{-6}).

Noncancer hazard is characterized by calculating an HQ based on the maximum one-year ADD for ingestion exposures and the RfD. The ADD is used instead of the LADD for non-carcinogenic endpoints because at least one of the co-critical effects identified for PFOA and PFOS is a developmental endpoint and can potentially result from a short-term exposure during critical periods of development. Unlike cancer risk estimates, HQs are risk indicators rather than risk estimates; the RfD represents a daily exposure that is likely to be without appreciable risk of deleterious noncancer effects during a lifetime. An HQ of 1 is used to establish a threshold of concern for a specific health effect. An HQ greater than 1 indicates risk (US EPA, 1986; 2000a; 2024e).

Equation 4. Human Cancer Risk (unitless)

Ingestion Exposures

$$LADD = \frac{ADD \times ED \times EF}{AT \times 365\ day/y};\ Risk_{oral} = LADD \times CSF_{oral}$$

Name	Description	Value
ADD	Average daily dose (mg/kg-day)	Calculated
ED	Exposure duration (yr)	10 years
EF	Exposure frequency (day/yr)	350 days/year
AT	Averaging time (yr)	70 years
CSF_{oral}	Oral cancer slope factor $(mg/kg\text{-}day)^{-1}$	29,300 $(mg/kg/day)^{-1}$for PFOA; 39.5 $(mg/kg/day)^{-1}$ for PFOS

Equation 5. Human Hazard Quotient, HQ (unitless)

Ingestion Exposures

$$HQ_{oral} = \frac{ADD}{RfD}$$

Name	Description	Value
ADD	Average daily dose (mg/kg-day)	Calculated
RfD	Noncancer reference dose (mg/kg-day)	3×10^{-8} mg/kg/day for PFOA; 1×10^{-7} mg/kg/day for PFOS

The following tables include CRLs and HQs from exposure to various media for PFOA and PFOS under each conceptual modeling scenario. All highlighted cells represent settings with risks or hazards exceeding the acceptable threshold. Risks are presented individually per pathway; a given receptor may have exposure from multiple pathways at one time. A given receptor may also have exposure to PFOA and PFOS at the same time. The presented risks and hazard quotients only represent risks contributed by contaminated biosolids use, not total risks to the receptor from that pathway, which may be larger.

4.2 Crop Farm Risk Estimation

The following table includes cancer risk levels and hazard quotients for receptors in the crop farm scenario, disaggregated by pathway.

Table 34. PFOA Risk Results for Crop Farm, Cancer and Non-Cancer

	Low Koc				High Koc			
	Adult		Child		Adult		Child	
Pathway	CRL	HQ	CRL*	HQ	CRL	HQ	CRL*	HQ
Dry Climate								
Exposed fruit	4.5E-06	0.039	4.6E-06	0.04	1.0E-05	0.09	1.0E-05	0.092
Exposed vegetable	3.7E-05	0.32	2.6E-05	0.23	8.5E-05	0.75	6.1E-05	0.53
Fish	4.9E-04	4.2	5.8E-04	4.9	9.1E-05	0.76	1.1E-04	0.89
Groundwater	2.1E-05	0.2	2.2E-05	0.21	1.1E-13	9.2E-10	1.2E-13	9.6E-10
Protected fruit	6.3E-06	0.054	6.9E-06	0.059	1.4E-05	0.13	1.6E-05	0.14
Protected vegetable	3.0E-05	0.26	5.5E-05	0.48	6.9E-05	0.61	1.3E-04	1.1
Root vegetable	2.2E-05	0.19	1.4E-05	0.12	4.9E-05	0.43	3.3E-05	0.29
Soil	1.7E-08	0.00018	2.6E-07	0.0027	4.6E-08	0.00042	7.1E-07	0.0064
Surface water	4.2E-04	3.6	4.4E-04	3.7	7.7E-05	0.65	8.1E-05	0.68
Moderate Climate								
Exposed fruit	2.6E-07	0.0033	2.7E-07	0.0034	4.2E-06	0.037	4.3E-06	0.038
Exposed vegetable	2.2E-06	0.027	1.6E-06	0.019	3.5E-05	0.31	2.5E-05	0.22
Fish	2.7E-05	0.24	3.1E-05	0.28	9.1E-05	0.79	1.1E-04	0.93
Groundwater	2.4E-04	2.5	2.5E-04	2.6	6.6E-06	0.055	6.9E-06	0.057
Protected fruit	3.7E-07	0.0046	4.0E-07	0.005	5.9E-06	0.052	6.5E-06	0.057
Protected vegetable	1.8E-06	0.022	3.3E-06	0.041	2.8E-05	0.25	5.2E-05	0.46
Root vegetable	1.3E-06	0.016	8.5E-07	0.011	2.0E-05	0.18	1.4E-05	0.12
Soil	2.8E-10	7.4E-06	4.3E-09	0.00011	1.5E-08	0.00016	2.2E-07	0.0025
Surface water	2.3E-05	0.2	2.4E-05	0.21	7.7E-05	0.67	8.1E-05	0.7
Wet Climate								
Exposed fruit	2.4E-07	0.0038	2.4E-07	0.0039	3.2E-06	0.028	3.3E-06	0.028
Exposed vegetable	2.0E-06	0.031	1.4E-06	0.022	2.7E-05	0.23	1.9E-05	0.16
Fish	1.8E-05	0.19	2.1E-05	0.22	6.4E-05	0.57	7.5E-05	0.66
Groundwater	2.4E-04	2	2.5E-04	2.1	2.6E-05	0.21	2.7E-05	0.22
Protected fruit	3.3E-07	0.0053	3.7E-07	0.0058	4.5E-06	0.039	4.9E-06	0.043
Protected vegetable	1.6E-06	0.026	3.0E-06	0.047	2.2E-05	0.19	4.0E-05	0.34
Root vegetable	1.1E-06	0.018	7.7E-07	0.012	1.6E-05	0.13	1.0E-05	0.09
Soil	2.6E-10	6.7E-06	4.0E-09	0.0001	1.0E-08	0.00011	1.6E-07	0.0017
Surface water	1.5E-05	0.16	1.6E-05	0.17	5.4E-05	0.48	5.7E-05	0.5

*CRLs for children represent lifetime cancer risks stemming from 10 years of exposure during childhood. These results do not describe risks of childhood cancer.

Table 35. PFOS Risk Results for Crop Farm, Cancer and Non-Cancer

	Low Koc				High Koc			
	Adult		Child		Adult		Child	
Pathway	CRL	HQ	CRL*	HQ	CRL	HQ	CRL*	HQ
Dry Climate								
Exposed fruit	2.3E-09	0.0044	2.4E-09	0.0045	4.0E-09	0.0076	4.1E-09	0.0077
Exposed vegetable	6.1E-09	0.012	4.4E-09	0.0083	1.0E-08	0.02	7.5E-09	0.014

Pathway	Low Koc				High Koc			
	Adult		Child		Adult		Child	
	CRL	HQ	CRL*	HQ	CRL	HQ	CRL*	HQ
Fish	1.1E-05	21	1.3E-05	25	1.0E-07	0.19	1.2E-07	0.22
Groundwater	3.9E-09	0.0078	4.1E-09	0.0082	2.9E-38	5.4E-32	3.1E-38	5.7E-32
Protected fruit	3.2E-09	0.0061	3.6E-09	0.0067	5.6E-09	0.011	6.1E-09	0.012
Protected vegetable	5.0E-09	0.0094	9.1E-09	0.017	8.5E-09	0.016	1.6E-08	0.030
Root vegetable	5.3E-08	0.1	3.6E-08	0.067	9.1E-08	0.17	6.1E-08	0.12
Soil	4.0E-11	8.8E-05	6.2E-10	0.0013	8.2E-11	0.00016	1.2E-09	0.0024
Surface water	3.3E-07	0.62	3.5E-07	0.65	2.8E-09	0.0053	3.0E-09	0.0055
Moderate Climate								
Exposed fruit	5.3E-10	0.0011	5.5E-10	0.0011	3.5E-09	0.0069	3.6E-09	0.007
Exposed vegetable	1.4E-09	0.0028	1.0E-09	0.002	9.3E-09	0.018	6.7E-09	0.013
Fish	4.3E-06	8.3	5.0E-06	9.7	1.3E-07	0.25	1.5E-07	0.29
Groundwater	7.1E-08	0.13	7.4E-08	0.14	1.3E-12	2.3E-06	1.3E-12	2.5E-06
Protected fruit	7.5E-10	0.0015	8.2E-10	0.0017	5.0E-09	0.0096	5.4E-09	0.011
Protected vegetable	1.1E-09	0.0023	2.1E-09	0.0042	7.6E-09	0.015	1.4E-08	0.027
Root vegetable	1.2E-08	0.025	8.2E-09	0.016	8.1E-08	0.16	5.4E-08	0.11
Soil	5.7E-12	1.8E-05	8.7E-11	0.00027	5.6E-11	0.00014	8.5E-10	0.0021
Surface water	1.3E-07	0.24	1.3E-07	0.25	3.5E-09	0.0068	3.6E-09	0.0071
Wet Climate								
Exposed fruit	3.9E-10	0.00081	3.9E-10	0.00083	3.2E-09	0.0062	3.3E-09	0.0063
Exposed vegetable	1.0E-09	0.0021	7.3E-10	0.0015	8.5E-09	0.016	6.0E-09	0.012
Fish	2.7E-06	5.2	3.1E-06	6.1	1.4E-07	0.27	1.6E-07	0.31
Groundwater	1.9E-07	0.36	2.0E-07	0.37	7.5E-10	0.002	7.9E-10	0.0021
Protected fruit	5.4E-10	0.0011	5.9E-10	0.0012	4.5E-09	0.0086	4.9E-09	0.0095
Protected vegetable	8.3E-10	0.0017	1.5E-09	0.0032	6.9E-09	0.013	1.3E-08	0.024
Root vegetable	8.8E-09	0.019	5.9E-09	0.012	7.4E-08	0.14	4.9E-08	0.095
Soil	3.3E-12	1.1E-05	5.1E-11	0.00017	5.6E-11	0.00012	8.6E-10	0.0019
Surface water	7.8E-08	0.15	8.2E-08	0.16	3.8E-09	0.0074	4.0E-09	0.0077

*CRLs for children represent lifetime cancer risks stemming from 10 years of exposure during childhood. These results do not describe risks of childhood cancer.

All highlighted cells represent hazard or cancer risk above acceptable thresholds for the crop farm scenario. Overall, PFOA risks are higher than those posed by PFOS and occur under more settings (low and high K_{oc}; dry, moderate, and wet climate conditions). The pathway with the highest risk for PFOS is fish consumption (CRL up to 1.3×10^{-5} and HQ up to 25). The pathways with the highest risks for PFOA are groundwater used as drinking water, surface water used as drinking water, and fish consumption, which have maximum cancer risks from 2.5×10^{-4} to 5.0×10^{-4} and maximum hazard quotients from 2.6 to 4.9. Every setting (dry, moderate, wet, low K_{oc}, high K_{oc}) results in at least one exceedance of cancer or hazard thresholds for every pathway. In the model, crop exposures result in cancer risk for PFOA, but these risks are based on greenhouse studies of pots in plants that likely over-estimate plant uptake and the estimates for plant uptake. See section 5.2 for more discussion of the uncertainties with uptake of PFOA and PFOS into fruits and vegetables

Soil concentrations remain below risk thresholds in all scenarios for PFOA and PFOS, but some scenarios are within a factor of 10 of the risk threshold. Notably, the only pathway exceeding risk thresholds for PFOS is fish consumption, and only when K_{oc} is low. This indicates that if soil sorption conditions are high for PFOS and only PFOS is present at low concentrations in biosolids, the material could be land applied for growing crops for human consumption without meaningfully increasing risk in any pathway. If land application occurs with a larger than 10-meter buffer from the closest fishable waterbody, this could mitigate risks posed by PFOS.

4.3 Pasture Farm Risk Estimation

The following table includes cancer risk levels and hazard quotients for receptors in the pasture farm scenario, disaggregated by pathway.

Table 36. PFOA Risk Results for Pasture Farm, Cancer and Non-Cancer

	Low Koc				High Koc			
	Adult		Child		Adult		Child	
Pathway	**CRL**	**HQ**	**CRL***	**HQ**	**CRL**	**HQ**	**CRL***	**HQ**
Dry Climate								
Beef	3.4E-05	0.41	4.4E-05	0.54	2.0E-04	1.7	2.6E-04	2.2
Eggs	7.7E-05	0.96	7.7E-05	0.96	6.3E-04	5.5	6.3E-04	5.5
Fish	6.3E-04	5.4	7.4E-04	6.3	2.7E-04	2.2	3.1E-04	2.6
Groundwater	1.5E-04	1.3	1.6E-04	1.3	1.4E-06	0.012	1.5E-06	0.012
Milk	4.1E-04	5	7.4E-04	9.1	2.1E-03	18	3.9E-03	34
Poultry	2.8E-06	0.035	5.1E-06	0.064	2.3E-05	0.2	4.2E-05	0.36
Soil	3.0E-08	0.00043	4.6E-07	0.0065	3.8E-07	0.0033	5.8E-06	0.05
Surface water	5.4E-04	4.6	5.6E-04	4.8	2.3E-04	1.9	2.4E-04	2
Moderate Climate								
Beef	6.5E-06	0.07	8.6E-06	0.092	2.7E-05	0.3	3.6E-05	0.4
Eggs	1.2E-05	0.15	1.2E-05	0.15	8.5E-05	0.97	8.5E-05	0.97
Fish	1.1E-04	1	1.3E-04	1.2	9.3E-05	0.81	1.1E-04	0.95
Groundwater	2.3E-04	1.9	2.4E-04	2	1.4E-05	0.12	1.5E-05	0.12
Milk	8.2E-05	0.86	1.5E-04	1.6	3.0E-04	3.3	5.5E-04	6.1
Poultry	4.3E-07	0.0055	7.9E-07	0.01	3.1E-06	0.035	5.7E-06	0.064
Soil	2.4E-09	5.1E-05	3.7E-08	0.00078	5.0E-08	0.00058	7.7E-07	0.0089
Surface water	9.6E-05	0.85	1.0E-04	0.89	7.9E-05	0.68	8.3E-05	0.71
Wet Climate								
Beef	4.5E-06	0.047	5.9E-06	0.061	1.9E-05	0.22	2.5E-05	0.29
Eggs	8.1E-06	0.098	8.1E-06	0.098	5.7E-05	0.69	5.7E-05	0.69
Fish	6.9E-05	0.61	8.1E-05	0.72	5.4E-05	0.46	6.4E-05	0.54
Groundwater	1.4E-04	1.2	1.4E-04	1.2	4.2E-05	0.35	4.4E-05	0.36
Milk	5.7E-05	0.57	1.0E-04	1	2.1E-04	2.5	3.8E-04	4.5
Poultry	2.9E-07	0.0036	5.4E-07	0.0065	2.1E-06	0.025	3.8E-06	0.046
Soil	1.5E-09	3.2E-05	2.3E-08	0.00049	3.3E-08	0.0004	5.0E-07	0.0061
Surface water	5.9E-05	0.52	6.1E-05	0.55	4.6E-05	0.39	4.8E-05	0.41

*CRLs for children represent lifetime cancer risks stemming from 10 years of exposure during childhood. These results do not describe risks of childhood cancer.

Table 37. PFOS Risk Results for Pasture Farm, Cancer and Non-Cancer

	Low Koc				High Koc			
	Adult		Child		Adult		Child	
Pathway	**CRL**	**HQ**	**CRL***	**HQ**	**CRL**	**HQ**	**CRL***	**HQ**
Dry Climate								
Beef	1.0E-06	2.3	1.4E-06	3	2.4E-06	4.7	3.2E-06	6.1
Eggs	6.2E-07	1.4	6.2E-07	1.4	2.1E-06	4	2.1E-06	4
Fish	2.1E-05	39	2.4E-05	45	6.2E-07	1.1	7.3E-07	1.3
Groundwater	1.6E-08	0.03	1.7E-08	0.031	4.4E-38	8.2E-32	4.6E-38	8.6E-32
Milk	1.3E-06	2.9	2.3E-06	5.3	2.6E-06	5	4.7E-06	9.1
Poultry	1.0E-07	0.23	1.9E-07	0.41	3.4E-07	0.66	6.2E-07	1.2
Soil	1.9E-10	0.00043	2.9E-09	0.0066	7.4E-10	0.0014	1.1E-08	0.022
Surface water	6.0E-07	1.1	6.3E-07	1.2	1.7E-08	0.032	1.8E-08	0.033
Moderate Climate								
Beef	2.5E-07	0.52	3.3E-07	0.69	1.5E-06	2.9	2.0E-06	3.8
Eggs	9.3E-08	0.23	9.3E-08	0.23	1.3E-06	2.5	1.3E-06	2.5
Fish	6.0E-06	12	7.0E-06	14	5.8E-07	1.1	6.8E-07	1.3
Groundwater	8.2E-08	0.15	8.6E-08	0.16	4.9E-11	9.1E-05	5.1E-11	9.5E-05

Pathway	Low Koc				High Koc			
	Adult		Child		Adult		Child	
	CRL	HQ	CRL*	HQ	CRL	HQ	CRL*	HQ
Milk	3.3E-07	0.69	6.1E-07	1.3	1.6E-06	3.1	3.0E-06	5.7
Poultry	1.5E-08	0.038	2.8E-08	0.068	2.1E-07	0.41	3.8E-07	0.74
Soil	2.0E-11	5.7E-05	3.0E-10	0.00087	4.5E-10	0.00089	6.9E-09	0.013
Surface water	1.7E-07	0.34	1.8E-07	0.35	1.6E-08	0.03	1.7E-08	0.032
Wet Climate								
Beef	9.7E-08	0.34	1.3E-07	0.44	9.8E-07	1.9	1.3E-06	2.5
Eggs	4.9E-08	0.15	4.9E-08	0.15	8.2E-07	1.6	8.2E-07	1.6
Fish	3.4E-06	6.6	4.0E-06	7.7	4.2E-07	0.81	4.9E-07	0.95
Groundwater	1.4E-07	0.26	1.5E-07	0.27	8.5E-10	0.0016	8.9E-10	0.0016
Milk	1.2E-07	0.44	2.2E-07	0.8	1.0E-06	2	1.9E-06	3.7
Poultry	8.1E-09	0.025	1.5E-08	0.045	1.4E-07	0.26	2.5E-07	0.48
Soil	1.4E-11	4.3E-05	2.1E-10	0.00065	2.9E-10	0.00057	4.5E-09	0.0086
Surface water	1.0E-07	0.19	1.0E-07	0.2	1.2E-08	0.022	1.2E-08	0.023

*CRLs for children represent lifetime cancer risks stemming from 10 years of exposure during childhood. These results do not describe risks of childhood cancer.

Modeling for the pasture farm scenario includes multiple pathways that exceed acceptable cancer risk or hazard levels for PFOA and PFOS. As is seen in the results from the crop farm scenario, PFOA results in more elevated risk pathways and pathways with higher risks than PFOS, due to PFOA's higher cancer slope factor. The highest risk pathways for PFOA include milk, beef, egg, fish, and drinking water consumption (either sourced from groundwater or surface water). The highest risk pathways for PFOS include fish, milk, eggs, and beef.

For PFOA, all climate and K_{oc} settings result in exceedances of acceptable risk and hazard thresholds for levels in milk, with cancer risk levels ranging from 5.7 x 10^{-5} to 3.9 x 10^{-3} and hazard quotients ranging from 5.7 to 34. The modeling suggests that even when modeled concentrations are below currently available method detection limits (MDLs), estimated cancer risks associated with PFOA can exceed acceptable thresholds. This indicates that there may be exceedances of acceptable risk thresholds due to PFOA levels in milk from farms with biosolids land application that fall below detectable limits. In the model, PFOS also exceeds risk thresholds in milk in most settings.

4.4 Reclamation Risk Estimation

The following table includes cancer risk levels and hazard quotients for receptors in the land reclamation scenario, disaggregated by pathway.

Table 38. PFOA Risk Results for Reclamation Site, Cancer and Non-Cancer

Pathway	Low Koc				High Koc			
	Adult		Child		Adult		Child	
	CRL	HQ	CRL*	HQ	CRL	HQ	CRL*	HQ
Dry Climate								
Beef	2.5E-05	0.39	3.2E-05	0.51	4.5E-05	0.41	5.9E-05	0.54
Eggs	7.4E-05	1.3	7.4E-05	1.3	1.5E-04	1.3	1.5E-04	1.3
Fish	1.1E-04	0.9	1.3E-04	1.1	3.5E-05	0.29	4.0E-05	0.34
Groundwater	9.4E-06	0.078	9.8E-06	0.081	1.6E-07	0.0013	1.7E-07	0.0014
Milk	2.7E-04	4.2	5.0E-04	7.7	4.9E-04	4.4	9.0E-04	8
Poultry	2.7E-06	0.048	4.9E-06	0.087	5.4E-06	0.049	9.8E-06	0.09
Soil	4.2E-08	0.00083	6.4E-07	0.013	9.0E-08	0.00084	1.4E-06	0.013
Surface water	9.1E-05	0.77	9.5E-05	0.8	2.9E-05	0.25	3.1E-05	0.26
Moderate Climate								
Beef	5.1E-06	0.25	6.7E-06	0.33	3.7E-05	0.4	4.8E-05	0.52
Eggs	1.6E-05	0.83	1.6E-05	0.83	1.1E-04	1.3	1.1E-04	1.3
Fish	1.2E-05	0.14	1.4E-05	0.16	2.7E-05	0.24	3.1E-05	0.28

Pathway	Low Koc				High Koc			
	Adult		Child		Adult		Child	
	CRL	HQ	CRL*	HQ	CRL	HQ	CRL*	HQ
Groundwater	2.9E-06	0.19	3.1E-06	0.2	1.2E-06	0.01	1.3E-06	0.011
Milk	5.7E-05	2.7	1.0E-04	5	4.0E-04	4.3	7.3E-04	7.9
Poultry	5.7E-07	0.03	1.0E-06	0.055	4.1E-06	0.048	7.5E-06	0.086
Soil	8.9E-09	0.00051	1.4E-07	0.0078	6.5E-08	0.0008	9.9E-07	0.012
Surface water	1.0E-05	0.12	1.1E-05	0.12	2.3E-05	0.2	2.4E-05	0.21
Wet Climate								
Beef	2.2E-06	0.11	2.9E-06	0.14	2.9E-05	0.36	3.9E-05	0.47
Eggs	5.6E-06	0.34	5.6E-06	0.34	8.7E-05	1.1	8.7E-05	1.1
Fish	1.6E-05	0.23	1.8E-05	0.27	2.7E-05	0.25	3.1E-05	0.3
Groundwater	1.3E-05	1.1	1.4E-05	1.1	1.7E-06	0.014	1.8E-06	0.015
Milk	2.6E-05	1.2	4.8E-05	2.2	3.2E-04	3.9	5.9E-04	7.1
Poultry	2.0E-07	0.012	3.7E-07	0.023	3.2E-06	0.042	5.8E-06	0.076
Soil	2.6E-09	0.0002	3.9E-08	0.0031	4.9E-08	0.00068	7.4E-07	0.01
Surface water	1.3E-05	0.2	1.4E-05	0.21	2.3E-05	0.22	2.4E-05	0.23

*CRLs for children represent lifetime cancer risks stemming from 10 years of exposure during childhood. These results do not describe risks of childhood cancer.

Table 39. PFOS Risk Results for Reclamation Site, Cancer and Non-Cancer

Pathway	Low Koc				High Koc			
	Adult		Child		Adult		Child	
	CRL	HQ	CRL*	HQ	CRL	HQ	CRL*	HQ
Dry Climate								
Beef	3.9E-07	0.82	5.1E-07	1.1	4.1E-07	0.82	5.4E-07	1.1
Eggs	3.1E-07	0.7	3.1E-07	0.7	3.5E-07	0.7	3.5E-07	0.7
Fish	3.1E-06	5.9	3.7E-06	6.9	8.3E-08	0.16	9.7E-08	0.18
Groundwater	2.3E-09	0.0042	2.4E-09	0.0044	1.5E-39	2.8E-33	1.6E-39	2.9E-33
Milk	4.3E-07	0.87	7.8E-07	1.6	4.4E-07	0.87	8.1E-07	1.6
Poultry	5.2E-08	0.11	9.4E-08	0.21	5.7E-08	0.12	1.0E-07	0.21
Soil	1.1E-10	0.00025	1.7E-09	0.0038	1.2E-10	0.00025	1.9E-09	0.0038
Surface water	9.2E-08	0.17	9.6E-08	0.18	2.3E-09	0.0044	2.4E-09	0.0046
Moderate Climate								
Beef	1.9E-07	0.76	2.5E-07	1	3.6E-07	0.81	4.8E-07	1.1
Eggs	1.3E-07	0.63	1.3E-07	0.63	2.9E-07	0.69	2.9E-07	0.69
Fish	1.2E-06	2.7	1.4E-06	3.2	7.7E-08	0.15	9.1E-08	0.17
Groundwater	9.6E-09	0.018	1.0E-08	0.019	8.0E-13	1.5E-06	8.4E-13	1.5E-06
Milk	2.2E-07	0.82	4.0E-07	1.5	3.9E-07	0.85	7.2E-07	1.6
Poultry	2.1E-08	0.1	3.8E-08	0.19	4.8E-08	0.11	8.7E-08	0.21
Soil	4.2E-11	0.00022	6.4E-10	0.0034	1.0E-10	0.00025	1.6E-09	0.0037
Surface water	3.6E-08	0.079	3.7E-08	0.082	2.1E-09	0.0041	2.2E-09	0.0043
Wet Climate								
Beef	1.7E-07	0.65	2.2E-07	0.85	3.1E-07	0.74	4.1E-07	0.98
Eggs	9.9E-08	0.49	9.9E-08	0.49	2.3E-07	0.62	2.3E-07	0.62
Fish	1.7E-06	4	2.0E-06	4.7	7.2E-08	0.14	8.4E-08	0.17
Groundwater	8.3E-09	0.015	8.7E-09	0.016	2.2E-12	4.1E-06	2.3E-12	4.3E-06
Milk	2.0E-07	0.73	3.8E-07	1.3	3.4E-07	0.79	6.2E-07	1.5
Poultry	1.6E-08	0.081	3.0E-08	0.15	3.9E-08	0.1	7.0E-08	0.18
Soil	3.1E-11	0.00018	4.7E-10	0.0027	8.1E-11	0.00022	1.2E-09	0.0033
Surface water	4.9E-08	0.12	5.1E-08	0.12	2.0E-09	0.0039	2.1E-09	0.0041

*CRLs for children represent lifetime cancer risks stemming from 10 years of exposure during childhood. These results do not describe risks of childhood cancer.

Modeling for the reclamation scenario includes multiple pathways that exceed acceptable cancer risk or hazard levels for PFOA and PFOS. As is seen in the modeling results from the pasture farm scenario, PFOA results in more elevated risk pathways and pathways with higher risks than PFOS owing to PFOA's higher cancer slope factor. The highest risk pathways for PFOA include milk, beef, egg, and drinking water consumption (either sourced from groundwater or surface water). The highest risk pathways for PFOS include fish and milk. Overall, risk levels in the reclamation scenario are lower than risks in the pasture farm scenario due to the modeling being based on a one-time biosolids application, rather than ongoing applications.

Though the reclamation scenario presents fewer risks than the pasture farm scenario, for PFOA, all climate and K_{oc} settings still result in exceedances of acceptable risk and hazard thresholds for levels in milk, with cancer risk levels up to 9.0×10^{-4} and hazard quotients up to 8. If one assumes that the remediation site does not include a grazing pasture for dairy cows, the modeling still suggests that there are risks for PFOA in groundwater, surface water, and fish pathways and for PFOS in fish pathways. Given the linear relationship between the loading of PFOA and PFOS to the field and the calculated risks, the risk results for a scenario with a single application of sewage sludge at a rate of 10 DMT/ha (more typical of a median farming scenario rather than a land reclamation scenario) would be 1/5 of the values presented in tables 38 and 39. This indicates that there are scenarios and pathways that may exceed the EPA's acceptable risk thresholds after a single application of 10 DMT/ha given the modeling conditions.

4.5 Potential Impacts beyond the Farm Family

The media concentrations modeled in the pasture and crop farm scenarios are relevant to many potential receptors beyond the farm family. Because the modeling suggests that risk thresholds are exceeded for individual exposure pathways, a person or population exposed through only one pathway (like drinking water or milk consumption) could still have an increased risk of adverse health effects. Potential impacts outside the farm family are described by pathway below:

Surface water and fish pathways. A land application site where PFOA and PFOS concentrations in biosolids were higher than 1 ppb and further from the surface waterbody may have similar outcomes to the modeled surface water and fish tissue concentrations. Thus, it is possible that a significant fraction of biosolids land application sites could have elevated PFOA and PFOS concentrations in surface water and fish tissue. These impacts could include drinking water concentrations that exceed acceptable risk thresholds and significant exposure from eating fish. Populations with elevated fish consumption rates could have higher exposures than the population modeled in this assessment (farmers).

Milk pathway. This assessment is focused on milk consumption by people living on dairy farms, who have higher milk consumption rates than the general population (US EPA 2018b). General population milk consumers are likely to consume milk blended from multiple farms with or without a history of biosolids land application. In parts of the U.S. with active dairy farms, community members may purchase milk and dairy products directly from local farms, either by participating in a CSA, frequenting farm stands, or purchasing their milk and dairy from a farmers' market. Those regularly consuming products from a farm contaminated with PFOA or PFOS would likely be at a greater risk than the general population, which is mostly comprised of people consuming blended milk products from a diversity of sources.

Groundwater pathway. Once PFOA and PFOS enter groundwater after leaching from soil, they will migrate along with the path of groundwater movement. The size of a PFOA or PFOS groundwater plume depends on the amount of the chemicals deposited on land, the rate of groundwater flow, and the time that has passed since application of biosolids contaminated with PFOA and PFOS. Depending on site-specific circumstances, there could be many neighboring families to a crop farm or pasture farm with impacted groundwater wells. Additionally, should a farm field that previously accepted biosolids be developed into housing later, there could be ongoing groundwater contamination, leading to drinking water impacts.

Beef and chicken pathways. This draft risk assessment models the concentration and risks associated with eating meat from laying hens and, in the case of PFOA, lactating cows. These scenarios were selected because for PFOA and PFOS, there is no data available on uptake into broiler hens, which are more commonly raised for meat. Similarly, though there are muscle uptake data available for PFOS accumulation into cattle used for beef production, the only PFOA data available for uptake into cow muscle is from a study that included dairy cows. Many families and commercial farms cull (and consume or sell for consumption) laying hens and dairy cows after they cease to produce sufficient quantities of milk or eggs. However, most chicken and beef consumed in the U.S. is not from these types of animals; most chicken is sourced from faster-growing broiler hens and most beef is sourced from cows like Black Angus, Red Angus, and Herefords. These animals raised primarily for meat will have different uptake factors for PFOA and PFOS and different dietary intakes than the laying hens and lactating dairy cows. For this reason, there are uncertainties in PFOA and PFOS exposure for those in the general population who do not have backyard hens and (for PFOA) dairy cows that they may slaughter for food.

Fruits and vegetables. As discussed previously, there are considerable uncertainties regarding the concentration and risk calculations for fruit and vegetable pathways due to data limitations on the uptake of PFOA and PFOS into these types of plants. However, there are many populations outside of the farm family that could be impacted by contamination of fruits and vegetables. It is increasingly popular for fruit and vegetable farms to develop CSA programs, where participants receive weekly deliveries of produce from a single farm and use this produce as their primary fruit and vegetable source. It is also not uncommon for families to frequent a single nearby farm stand or farmers market stand as a primary source of produce, especially during the fall, summer, and spring seasons. Finally, there are many home gardeners who, for a hobby or for economic reasons, grow a large portion of their produce in their yard or at a community garden plot. Because these groups also primarily source their produce from a single site, should there be PFOA and PFOS biosolids impacts, produce could be a meaningful source of exposure.

4.6 Sewage Sludge Disposal Site Risk Estimation

The following table includes cancer risk levels and hazard quotients for drinking water receptors in the surface disposal scenario.

Table 40. PFOA Groundwater Risk Results for Sludge Disposal Site, Cancer and Non-Cancer

	Low Koc				High Koc			
	Adult		Child		Adult		Child	
Pathway	CRL	HQ	CRL*	HQ	CRL	HQ	CRL*	HQ
Dry Climate								
No liner	1.3E-03	11	1.4E-03	12	4.0E-06	0.034	4.2E-06	0.036
Clay liner	1.1E-03	9.2	1.2E-03	9.6	2.7E-06	0.023	2.8E-06	0.024
Composite liner	7.3E-07	0.0061	7.6E-07	0.0063	8.5E-35	7.2E-31	8.9E-35	7.5E-31
Moderate Climate								
No liner	4.8E-04	4	5.0E-04	4.2	1.3E-06	0.011	1.3E-06	0.011
Clay liner	3.1E-04	2.6	3.2E-04	2.7	8.4E-07	0.0071	8.7E-07	0.0074
Composite liner	5.9E-08	0.0005	6.2E-08	0.00052	8.0E-35	6.9E-31	8.4E-35	7.2E-31
Wet Climate								
No liner	8.5E-04	7.5	8.9E-04	7.8	9.0E-06	0.076	9.4E-06	0.08
Clay liner	6.6E-04	5.6	6.9E-04	5.9	4.1E-06	0.035	4.3E-06	0.037
Composite liner	2.2E-07	0.0018	2.3E-07	0.0019	4.7E-17	4E-13	4.9E-17	4.2E-13

*CRLs for children represent lifetime cancer risks stemming from 10 years of exposure during childhood. These results do not describe risks of childhood cancer.

Table 41. PFOS Groundwater Risk Results for Sludge Disposal Site, Cancer and Non-Cancer

Pathway	Low Koc				High Koc			
	Adult		Child		Adult		Child	
	CRL	HQ	CRL*	HQ	CRL	HQ	CRL*	HQ
Dry Climate								
No liner	9.5E-08	0.18	1.0E-07	0.19	3.3E-11	6.4E-05	3.5E-11	6.7E-05
Clay liner	6.6E-08	0.12	6.9E-08	0.13	2.3E-11	4.4E-05	2.4E-11	4.6E-05
Composite liner	1.6E-13	3.1E-07	1.7E-13	3.2E-07	1.4E-39	2.7E-33	1.5E-39	2.9E-33
Moderate Climate								
No liner	3.1E-08	0.059	3.3E-08	0.062	1.3E-11	2.5E-05	1.3E-11	2.6E-05
Clay liner	1.8E-08	0.034	1.9E-08	0.035	7.3E-12	1.4E-05	7.6E-12	1.5E-05
Composite liner	3.3E-21	6.2E-15	3.4E-21	6.5E-15	1.6E-39	3.1E-33	1.6E-39	3.2E-33
Wet Climate								
No liner	1.6E-07	0.29	1.6E-07	0.31	1.6E-10	0.00031	1.6E-10	0.00032
Clay liner	8.4E-08	0.16	8.8E-08	0.16	6.7E-11	0.00013	7.0E-11	0.00014
Composite liner	9.0E-13	1.7E-06	9.4E-13	1.8E-06	2.3E-39	4.5E-33	2.4E-39	4.7E-33

*CRLs for children represent lifetime cancer risks stemming from 10 years of exposure during childhood. These results do not describe risks of childhood cancer.

The surface disposal scenario models groundwater impacts with three types of liner options: no liner, clay liner, and composite liner. The modeling runs suggest that for PFOA, cancer risk thresholds are exceeded in all scenarios where the surface disposal site is unlined or lined with clay. For unlined surface disposal sites, cancer risks for PFOA in groundwater range from 1.3 x 10^{-6} to 1.4 x 10^{-3}. The upper end of these values represents risks three orders of magnitude higher than the acceptable threshold. Hazard quotients in this setting for PFOA go up to 12 in dry climates for child receptors. Unlined surface disposal sites and lagoons are common across the U.S., and thus groundwater around these sites is at high risk for contamination. Risks are only slightly mitigated by using a clay liner, and but are significantly mitigated by the use of a composite liner.

Unlike PFOA, PFOS appears to be less mobile in surface disposal sites and therefore poses lower risks. PFOS also has less potent toxicity than PFOA, such that concentrations can be higher without exceeding risk thresholds. None of the surface disposal lining options result in exceedances of PFOS risk thresholds when the concentration of sludge is 1 ppb. Given that there is a linear relationship between the starting concentration of PFOS in sludge and the groundwater concentration, it is anticipated that cancer risk thresholds may be exceeded in some unlined scenarios around concentrations of 10 ppb for PFOS and HQs may exceed 1 at concentrations around 4 ppb in some unlined scenarios and 5 ppb in some clay-lined scenarios.

4.7 Other Land Application Risk Estimation

As described in Section 2.8, there are many biosolids land application scenarios that are not quantitatively or qualitatively assessed in this document. Examples include land application of biosolids or septage on turf fields, golf courses, tree farms, or natural forested areas. Based on the risk values for pathways like groundwater, surface water, fish, and soil of the farming scenarios, it is possible that application of biosolids or septage in these alternative land application scenarios could also lead to exceedances of acceptable risk thresholds in these pathways. For PFOS, runoff from an 80-acre application site to a 13-acre lake or reservoir could result in risk exceedances for fish and surface water pathways – a typical 18-hole golf course requires 100 to 175 acres of land. For PFOA, applications of this size could result in risk exceedances for groundwater, surface water, and fish pathways as well. This indicates that ongoing use of biosolids to fertilize a golf course could present risks, especially if there are nearby water bodies used for fishing or drinking water or if there were downgradient residents using groundwater as a source of drinking water.

When biosolids are used to fertilize forested lands, there may or may not be ongoing annual applications of biosolids. However, for the remediation pathway models, one-time application of 1 ppb PFOA and PFOS in biosolids still results in exceedances of groundwater, surface water, and fish risk thresholds in most climate and sorption scenarios. This indicates that even one-time application of biosolids to a forested site could present risks, depending on the concentrations of PFOA and PFOS in the biosolids, the size of applied lands, the size of the nearby waterbody, and if there were any receptors nearby, such as those eating fish from waterways, those hunting or gathering food, those using groundwater as a source of drinking water, or those using surface water as a source of drinking water. Additionally, in some parts of the U.S., forests are used for livestock grazing. That said, there are many site-specific factors that could influence risk in forested settings, including the fate and transport behaviors in those specific settings.

Use of biosolids in road construction projects could present risks, depending on how the biosolids were used, the amount used, and the concentration of PFOA and PFOS in the materials. Application of biosolids as a groundcover over small areas of roadside likely represents much less land cover area than the 80-acre fields modeled in this assessment. However, depending on the conditions, the disturbed land adjacent to roadwork could present higher risks of transport through runoff and erosion to a nearby waterbody.

Application of Class A_{EQ} biosolids to residential areas (parks, schools, playgrounds, homes) could pose risks to children from incidental soil ingestion of biosolids or soil contaminated with PFOA or PFOS. The EPA has posted non-cancer residential soil screening levels for CERCLA site evaluation at 1.9 ppb and 6.3 ppb for PFOA and PFOS, respectively, based on non-cancer risks; soil levels based on cancer for PFOA are lower at 0.019 ppb (more stringent) (US EPA 2024i). The CERCLA screening levels are calculated with a soil ingestion rate (200 mg/day) based on upper percentile of behavior patterns of children presented in the Exposure Factors Handbook. The soil ingestion rate used in developing CERCLA soil screening levels is larger than the soil ingestion rate used in this assessment (40 mg/day), which is meant to represent central tendency exposures. The goal of this central tendency risk assessment is to identify the potential scope and magnitude of risks under different biosolids use and disposal scenarios; historically, EPA biosolids assessments have used upper percentile estimates to derive risk-based values, consistent with other EPA programs (US EPA 1992; US EPA 2003a).

4.8 Additional Risk Considerations for All Scenarios

This draft risk assessment is based on the simplification that the risk to human health from sewage sludge use or disposal can be represented by focusing on the concentrations of PFOS or PFOA in sewage sludge and the resulting soil or other media concentrations. However, studies of sewage sludge indicate that precursors to both PFOS and PFOA are present (see Sections 2.2.2 and 2.4) and several studies indicate that ongoing loading of PFOA and PFOS to soils occurs over time through the degradation of precursors that were also present in sewage sludge (see Section 2.2.2). Several of these precursors are also present in EPA Method 1633 and may be monitored with that method in soil, water, and sewage sludge. Concentrations of PFOA and PFOS would increase in each medium if precursors were included in this assessment, resulting in an increased risk finding. Precursors to PFOA and PFOS may also pose their own hazards to human health.

The risk tables in this assessment display results for adults; these risk values represent an average risk between women and men of adult age. The EPA's final toxicity assessments conclude that both PFOA and PFOS are likely to cause cancer, hepatic effects and cardiovascular effects; these effects are relevant to men and women in the adult population (US EPA 2024b;c). However, the EPA's toxicity assessments also conclude that PFOA and PFOS are likely to cause developmental effects in children when mothers are exposed during pregnancy or when the infants are exposed during early life. The development effects are listed as co-critical with the hepatic and cardiovascular effects, which indicates that they are equally sensitive effects as the other critical effects. Women of childbearing age, pregnant women, and lactating women all have elevated bodyweight-normalized drinking water intake rates compared to the general adult population (US EPA, 2019c). The median drinking water intake rate used for the general population in this assessment is 13.4 ml/kg-day. Though the median drinking water intake rates for women of childbearing age, pregnant women, and lactating women are not presented in the EFH, the mean drinking water consumption rates for those groups are as follows: 15.6 ml/kg-day for women of childbearing age, 15.5 ml/kg-day for pregnant women, and 22.9 ml/kg-day for lactating women (EFH, Chapter 3, Table 3-3; US EPA 2019c). These values are from 14-71% higher than the drinking water intake rates for the general population. For this reason, there are some drinking water pathways in the surface disposal scenario that are currently not exceeding the risk threshold for adults but would do so for lactating women.

4.9 Monte Carlo Analysis

The central tendency deterministic modeling described in this draft risk assessment suggests that there are unacceptable risks associated with PFOA and PFOS in multiple individual exposure pathways across every assessed use and disposal practice, even when central tendency exposure parameters are assumed. Further refinement of the risk assessment from the central tendency model to a probabilistic risk assessment would result in an increased risk finding because the goal of a probabilistic assessment is to identify the threshold protective of 95th percentile exposures, while the central tendency modeling is modeling median (50th percentile) conditions. For this reason, the EPA is not conducting additional modeling exercises at this time, but rather focusing on sharing the central tendency modeling results and identifying actions that could be taken to mitigate risks. Any further refinement of the draft risk assessment (*e.g.*, probabilistic modeling of 95th percentile exposures) would delay future risk management decisions.

5 UNCERTAINTY, VARIABILITY, AND SENSITIVITY

5.1 Variability

Variability describes the changes in true conditions for a parameter over time or space. Nearly every parameter used to run the biosolids use and disposal models are variable across U.S. populations or geography. For example, the meteorological data for each modeled climate scenario (dry, moderate wet) is variable over time and space. Soil composition can be variable regionally but may also vary within a single farm or site. Uptake factors for plants and livestock vary by species and location; human consumption of these plants and animal products also vary individually and by region. Though a Monte Carlo analysis would allow for the quantification and propagation of variability throughout the modeling process, the median risks presented at the lowest detectable PFOA and PFOS concentrations are high enough to ensure that modeling 95th percentile exposure scenarios – even when quantifying variability and uncertainty – would also result in unacceptable risk scenarios. In selecting median values for most of the input parameters, and selecting high *and* low values for the most sensitive parameters, the outputs represent a set of reasonable risk or hazard values that are relevant to the diversity of biosolids use and disposal sites in the U.S. Again, this assessment is not designed to capture site-specific conditions or outcomes, but rather give an estimate of the range of realistic outcomes that are possible across a variety of common scenarios that exist in the U.S. and inform potential future risk mitigation actions.

5.2 Uncertainty

There are two types of uncertainty: 1) systemic uncertainty, which are unknowns, errors, or assumptions that produce results in one direction, and 2) random uncertainty, which generates in a spread of final results above and below the central tendency value (*i.e.*, median value). An example of random uncertainty is uncertainty introduced by the modeling of the impacts of weather, where there are random natural variations in parameters like rainfall year to year. This assessment includes both systemic uncertainties and random uncertainties. Some systemic uncertainties produce results towards higher-risk outcomes, and some produce results towards lower risk outcomes.

5.2.1 Systemic Uncertainties Resulting in Underestimation of Risk

This assessment includes several assumptions that could result in an underestimate of risk at specific sites. Perhaps most significantly, this assessment assumes that the starting concentration of PFOA and PFOS in biosolids is only 1 ppb. The available biosolids monitoring data from the U.S. suggest that nearly all biosolids have higher concentrations than this threshold; for example, the annual average PFOS concentration in biosolids produced in Maine is between 16 ppb and 27 ppb from 2019-2022 and the annual average PFOA concentration is between 5.3 ppb and 9.4 ppb during this same time window (Maine DEP, 2023). Sampling from other states (Michigan, California) align with these trends (Link et al., 2024; Mendez et al., 2021). Highly impacted biosolids can exceed 10 times the average concentrations (Higgins et al., 2005; 3M, 2001). Furthermore, the modeling indicates that PFOA and PFOS incorporated into soils from biosolids can be persistent sources of contamination to groundwater, surface water, and human or animal food over time; concentrations of PFOA and PFOS in the past were likely higher than currently observed due to the historically high use of PFOA and PFOS in commercial, industrial, and consumer products. In this way, historic land application of contaminated biosolids could present significantly more ongoing risks than current-day applications.

A second significant systemic uncertainty that underestimates risk in this assessment is that PFOA and PFOS precursors cannot be included in the model at this time due incomplete information about which PFOA and PFOS precursors are present in sewage sludge, the rate of transformation of each precursor to its terminal degradate, the yield of PFOA and PFOS generation, the toxicity of the precursors, and the environmental fate of the precursors. As discussed previously, PFOA and PFOS precursors are well known to act as ongoing sources of PFOA and PFOS in soils, like FTOHs and diPAPs. Some precursors are measurable using EPA Method 1633, yet others are not. Basing a risk assessment solely on the presence of PFOA and PFOS will therefore result in modeling that underestimates the exposures and risks resulting from reuse or disposal of biosolids because this assessment is not accounting for additional loading of PFOA and PFOS over time as precursors transform. The EPA may consider whether the environmental precursors for PFOA and PFOS should be included in the future.

A third systemic uncertainty that results in an underestimate of risk are assumptions in the models related to each receptor’s exposure outside a single residence. Currently the models assume that there are zero exposures to PFOA and PFOS during the times when someone is traveling away from their home and during the majority of the years of their life when they are not living at the impacted site (60 of their 70 years of life are assumed to have zero PFOA and PFOS exposure from any source). The EPA’s National Primary Drinking Water Regulation for PFAS (April 26, 2024; 89 FR 32532) estimates that 6-10% of all public drinking water systems in the US contain detectable amounts of PFOA and PFOS (the prevalence of PFOA and PFOS contamination in private groundwater wells is not known). It is also known that there are many other pervasive sources of PFOA and PFOS exposure that are unrelated to biosolids use and disposal (*e.g.*, foods like fish and shellfish; consumer products; household dust). It is likely that even if a person moved from a residence impacted by PFOA and PFOS contamination from a biosolids-related source, they would still have ongoing sources of PFOA and PFOS exposure. This assessment also does not attempt to estimate the concentrations of these chemicals that may occur in human breastmilk due to sewage sludge related exposures or non-sewage sludge related exposures. Therefore, readers should interpret risk estimates for each pathway narrowly as risk stemming from the biosolids use or disposal only, and not total risk to the receptor.

A fourth systemic uncertainty that results in underestimation of risk are the assumptions that no “background” levels of PFOA or PFOS are present in soil from long-range atmospheric transport of PFOA, PFOS, and their precursors or any other source of non-biosolids related contamination to the farm. Rankin et al. 2016 sampled soils across North America and the globe that were judged to have “no evident human impact,” meaning that they were from undeveloped locations with no known or likely proximal point sources of PFAS. PFOA was detected in all soil samples, and PFOS was detected in all samples except one from rural Estonia. Even the most remote samples included in this study (locations like Lake Bonney, Antarctica; Mapunguwe National Park, South Africa; Inuvik, Canada; and Montevideo, Uruguay) had measurable levels of PFOA from 15 to 270 ppt and PFOS from 4 to 26 ppt. Though it is possible that there were unknown local sources of PFOA and PFOS to these soils, it is likely that some amount of PFOA and PFOS are present ubiquitously around the globe and the US. These background soil concentrations are within the range of modeled soil results for land application of biosolids containing PFOA and PFOS at 1 ppb, especially in low K_{oc} settings. If contributions of PFOA and PFOS were considered from ongoing and historic atmospheric deposition, risks and hazards in these pathways would increase. Again, readers should interpret the risks presented in this draft risk assessment as added risks solely from sewage sludge use or disposal, not total risks to the receptor.

Finally, this draft risk assessment does not attempt to quantify total (aggregate) exposures or risks to a single receptor to each chemical, nor does it account for PFOA and PFOS dose additivity. Aggregate exposure and risk assessment involve the analysis of exposure to a single chemical by multiple pathways and routes of exposure. This assessment does not aggregate exposure and risk, and instead presents estimated exposure and risk for each individual exposure pathway that was modeled (*i.e.*, consuming fish, drinking water, incidentally ingesting soil). This approach does not account for exposure from multiple modeled pathways simultaneously, sewage sludge-related pathways that were not modeled due to data gaps (including inhalation and dermal exposure pathways) or exposure pathways not related to sewage sludge use and disposal (such as exposure from use of personal care products, cleaning supplies, household dust, etc.).

This decision to assess each pathway individually allows modeling results to be interpreted as risk contributed from sewage sludge for each pathway across a variety of sewage sludge use and disposal scenarios. However, in each given scenario, a receptor may be exposed from multiple pathways at the same time as well as via pathways not modeled in this draft risk assessment. For example, farmers who consume animal products produced on the farm likely also consume drinking water sourced locally as many rural areas of the country rely on groundwater. That farmer may also have PFOA or PFOS exposure that is unrelated to the land application of biosolids on his property. Other farm families with biosolids land application on their property may be largely self-sufficient, sourcing nearly all of their produce, animal products, and water from their property. These families would have biosolids-related exposures from many or all the modeled pathways. Still more individuals may be impacted by a single pathway of biosolids-related exposures, such as a person who fishes from an impacted waterbody but has no other sources of biosolids-related exposures, or an individual whose drinking water source is impacted, but otherwise sources food from non-impacted sources. These pathways are not summed in the assessment and outside exposure is not accounted for using a relative source contribution (RSC) term or any other method. There is a substantial amount of variability and uncertainty surrounding the populations who are exposed to one or multiple pathways of biosolids-related exposure. Because single pathways of exposure may result in exceedances of acceptable risk levels and because there are unknowns regarding the numbers of people exposed to each combination of exposure pathways, the EPA finds that presenting pathway-specific risks is the most efficient way of presenting risks at this time.

Cumulative exposure and risk assessment involve analysis of exposures from multiple stressors that occur simultaneously. A receptor may be exposed to both PFOA and PFOS at the same time. PFOA and PFOS have been shown to be dose additive (US EPA, 2024e) and are nearly always found in mixtures in biosolids, and it follows that the environmental media impacted by use or disposal of biosolids also contains mixtures of PFOA and PFOS. The presence of mixtures and multiple pathways for exposure would result in higher risks of adverse health effects at a population scale than are reflected in the pathway-specific results.

5.2.2 Systemic Uncertainties that Result in Overestimation of Risk

One systemic uncertainty resulting in an overestimate of risk stems from a lack of data on PFOA and PFOS plant uptake factors. There are limited studies of uptake into fruits and vegetables, particularly in field conditions where biosolids are a source of contamination. Studies of biosolids-amended soils generally find less uptake under field conditions than when the same biosolids-amended soils are moved to a pot and plants are cultivated in a greenhouse. Though this assessment aimed to use biosolids-specific field studies for parameterizing vegetable and fruit uptake, there were no such studies available. Based on the observed trend that field-based studies produce uptake values that are consistently lower than greenhouse studies, if there were more biosolids-specific field data available for the entire basket of often-grown fruits and vegetables in the U.S., the mean uptake factors may be lower than the one currently used in this assessment. It should be noted that the data for uptake into plants like grasses used for forage, hay, or silage did not have these same issues related to availability of field data, so livestock exposures are based on studies of plants in fields where biosolids were land applied.

Another systemic uncertainty resulting in an overestimate of risk is the inability to account for precursors presence when parameterizing uptake values for food crops, feed crops, and livestock. The available livestock studies, in particular, may be capturing contamination settings where precursors to PFOA and PFOS are available in addition to PFOA and PFOS themselves. If these precursors transform in the livestock to PFOA and PFOS due to metabolism in the liver or other organs, this will result in an overestimate of PFOA and PFOS uptake. There is more discussion of this potential effect in the livestock model parameterization section of the report (Section 2.9.3.5).

The current modeling scenario assumes that a farm will receive yearly applications of biosolids for 40 consecutive years, which is consistent with the prior EPA biosolids risk assessment for PCBs and dioxins (US EPA, 2003a) but lower than the years assumed to calculate the annual and cumulative loading rates for metals that support the existing sewage sludge regulations under CWA section 503. The use of 40 years may be an overestimate of the loading for some farms, but the EPA does not have data to indicate the frequency of application at a given site across the country. The current biosolids regulations allow land application to happen yearly or multiple times per year if the amount of biosolids land applied is consistent with the nitrogen needs of the crops grown at the farm, and thus, a 40 consecutive years of annual biosolids application on a farm is a reasonable assumption.

5.2.3 Random Uncertainties

Most of the random uncertainties included in this report stem from modeling parameters where there are data limitations, resulting in an over- or underestimation of the “true” conditions. For example, exposure factors used in this assessment (drinking water intake, fish intake, intake of various types of foods) are based on surveys conducted at various times in the U.S. These surveys vary in sample size and methodology and may be imperfect measurements of “true” consumption behavior. These surveys also do not capture all potentially relevant consumption behavior, like the consumption of animal livers, which are known to have higher levels of PFOA and PFOS than muscle tissues. As a result, the mean or median of the survey may be over- or underestimating reality. Despite these uncertainties, this assessment relies on the best available datasets for exposure factors.

Other random uncertainties are introduced in the three sites and two K_{oc} settings used in the fate and transport models. The EPA selected hydrogeological and geochemical conditions at three locations, using data from those sites to ensure that the combination of parameters at each site were as realistic as possible. Of course, there is a large amount of variability in the U.S. in site conditions, for example, variability in depth to groundwater. Though these three sites represent wet, moderate, and dry climates in the US, they may not capture the full extent of important hydrogeological and geochemical conditions. Any specific site with biosolids use or disposal may or may not be well-approximated by these models.

5.3 Sensitivity of Models

The EPA assessed the sensitivity of each model parameter in the groundwater and surface water models used in this report. Overall, the EPA finds that the K_{oc}, depth from ground surface to water table, hydraulic gradient in the aquifer, and hydraulic conductivity of saturated zone parameters are the most sensitive in the groundwater models. K_{oc} and f_{oc} are the most sensitive parameters in the surface water models. The detailed results of the sensitivity analysis can be found in Appendix D.

6 COMPARISON OF MODELED CONCENTRATIONS AND OBSERVED CONCENTRATIONS IN RELEVANT MEDIA

Though this draft risk assessment is not aiming to model risks stemming from biosolids use or disposal at any specific site, the modeled concentrations generated in this assessment seem reasonable when compared to “real life” observations of PFOA and PFOS in various media stemming from contamination of biosolids. The best datasets available for ground truthing our models would include known PFOA and PFOS composition of the land-applied biosolids, known timeframes for when the biosolids were applied and known application rates, observed concentrations of PFOA and PFOS in all of the relevant media (soil, groundwater, surface water, fish, produce, livestock feed, animal products), and a very detailed understanding of the hydrogeological and soil conditions at the site. To date, such a complete study is not available. However, there are other datasets with some of this information that can be used to determine if the range of modeled results in this assessment are supported by real-world observations. These datasets generally represent high-end contamination scenarios in the U.S. prior to the phase out of PFOA and PFOS, though there is one study of a field-based experiment in Ontario, Canada with mass loading rates of PFOA and PFOS that are more analogous to those used in this draft risk assessment. The high-end contamination scenarios are also useful in understanding the fate and transport behaviors of PFOA and PFOS in natural environments after land application of biosolids.

6.1 Biosolids Investigations in Ottawa, Ontario, Canada

In 2008, Canadian researchers applied dewatered municipal biosolids to a 14-hectare experimental research field located in Ottawa that had never previously had biosolids applied (Gottschall et al., 2017). The biosolids were applied one time at a rate of 22 Mg dry weight per hectare (equivalent to 22 MT dw/ha). The biosolids applied to the field contained 1.6 ng/g (ppb) PFOA and 7.2 ng/g (ppb) PFOS. The researchers then planted winter wheat and spring wheat on the field with biosolids application and a control plot in the same research station without any history of biosolids application. Both fields were independently tile drained (tile drains are an artificial subsurface drainage system installed to facilitate plant growth in wetter climates; these drainage systems are also commonly used in regions of the

United States). In this case, the tile drains were installed 1.2 meters below the soil surface. The researchers then monitored groundwater, tile drainage water, soil, and wheat grain.

The details of sampling strategies for these media are described in Gottschall et al. 2017. In brief, shallow soil samples (0-0.3 m depth) were collected in triplicate pre-application, 6 months, 9 months and 12 months post application from each of the 8 sampling locations in the biosolids applied plot and the reference plot. Due to the competent nature of the dewatered biosolids and the strategy used for their incorporation (mouldboard plowing), it was also possible to identify biosolids aggregates in the top layer of soils even a year after biosolids application. These aggregates were also sampled for PFAS content at 1 month, 2, months, 6 months, and 12 months post biosolids application. Tile water was sampled during rain events, with the first sample collected within 15 minutes of rain event water appearing in the drainage system, followed by sampling at 1, 2, and 6 hour intervals. Due to cost constraints, only the first sample was analyzed for PFAS concentrations because this sample was expected to have the highest concentration of pollutants. Some additional tile water samples were also collected during low flow conditions. For groundwater sampling, each field (control and experimental) had two piezometer nests/wells. Each piezometer nest included three piezometers with intakes centered at 2, 4, and 6 meters below the soil surface. Groundwater was sampled on a monthly basis pre- and post-application. Pre- and post-application groundwater samples were then pooled by depth for PFAS analysis. Grain was sampled from the harvester grain storage bin at various intervals during the harvesting process and mixed to form a single composite grain sample for each field; the reference field was harvested first to avoid cross contamination of grain samples. In total, the post-application monitoring period for this study spanned from October 2008 to November 2009.

Pre-application soils in the reference field and experimental field had low or non-detectable levels of PFOA and PFOS (PFOA of 118 ppt and non-detectable PFOS in the experimental field; ~100 ppt PFOA and PFOS in the reference field). In the biosolids application field, post-application soil samples had increasing levels of PFOA and PFOS throughout the study period. PFOA levels in surface soils increase from ~100 ppt before application to ~400 ppt at 6 months and ~800 ppt at 9 and 12 months. PFOS levels in surface soils increase from non-detectable to 200 ppt at 9 months and 400 ppt at 12 months (a 6-month concentration is not reported for this compound). This increase in soil PFOA and PFOS concentrations after a single biosolids application could be due to the slow release and mixing of biosolids aggregates into soils and/or degradation of PFOA and PFOS precursors, which were not measured in this study.

The soil concentrations in this field study are reasonably well-aligned with the modeled concentrations of PFOA and PFOS reclamation scenario of this assessment, though the slow breakdown of biosolids aggregates and the possible presence of PFOA and PFOS precursors are likely influencing the fate and transport of PFOA and PFOS in the field study. The reclamation scenario modeled in this assessment assumed an application rate of 50 Mt dry weight per hectare of biosolids containing 1 ppb PFOA and PFOS. This amounts to an application of 50 mg/ha of PFOA and PFOS. The Ontario study applied biosolids at a rate of 22 Mt/ha with a starting biosolids concentration of 1.6 ppb PFOA and 7.2 ppb PFOS, which amounts to an application of 3.52 mg/ha PFOA (~14 x lower than modeled) and 158 mg/ha PFOS (~3 x higher than modeled). Because our models assume a linear relationship between the PFOA and PFOS mass loading and the corresponding soil concentrations, our modeling would expect soil concentrations in this scenario to range from 0.4 to 14 ppt for PFOA and from 135-600 ppt PFOS. The measured soil values for PFOA (~800 ppt) are higher than what was expected by ~10-80 times, but the measured values for PFOS (~400 ppt) are within the range of expected results. The discrepancy between measured and modeled soil concentrations for PFOA could be due to PFOA precursors present in the field study biosolids, the challenges of sampling soils with heterogeneous inclusion of biosolids aggregates, or other site-specific factors.

The authors report PFOA and PFOS concentrations in tile drainage water and groundwater before and after the application of biosolids in the experimental and reference plots. For groundwater, the reference plot had “marginally detectable” (0.5-0.6 ppt) levels of PFOS at the end of the monitoring period, but no detectable levels of PFOA. In the experimental plot, PFOA was detected in groundwater after biosolids application, with concentrations ranging from 1.5-3 ppt over the course of the year. PFOS was also detected in groundwater after the application of biosolids to the experimental plot (0.8 ppt), but this detection did not occur until one year after the application of biosolids. For tile drainage samples, the reference plot had one detection of PFOS in tile drainage (~1.2 ppt) before the biosolids were applied to the experimental plot but had non-detectable levels of PFOS in tile drainage in all subsequent samples. There was no PFOS detected in tile drainage water at the experimental plot prior to biosolids application. There was also no PFOA detected in tile drainage water in any of the control plot samples or in the experimental plot prior to biosolids land application. The PFOS concentrations in post-application tile drainage water from the experimental plot were mostly non-detectable, but there was a sample with ~1.2 ppt PFOS shortly after the biosolids land application and a sample with ~0.5 ppt PFOS about six months following biosolids application. The PFOA concentrations in tile drainage water at the experimental plot after biosolids application were also mostly non-detectable, but there were three samples with detections that ranged from ~4 to 24 ppt.

The modeling in this assessment does not attempt to capture the potential effects of tile drainage on surface water or groundwater fate and transport dynamics for PFOA and PFOS. The modeling in this risk assessment is also predicting concentrations of PFOA and PFOS in a nearby surface water body (a lake or pond), which is not analogous to concentrations in tile drainage water. Finally, the low levels of PFOA and PFOS in biosolids applied in this study result in water media concentrations that are close to the detection limits for these compounds, which can render results difficult to interpret. That said, the trends in groundwater and tile drainage water concentrations observed in this study broadly align with trends observed in this assessment’s modeling of groundwater and surface water. Firstly, the researchers found consistently elevated PFOA concentrations in groundwater and occasionally elevated PFOA concentrations in tile drainage water. The fact that concentrations were higher and more frequently detected in groundwater and tile drainage water than PFOS aligns with the observation in our assessment that PFOA is more mobile in water than PFOS. Assuming a linear relationship between the mass loading and groundwater concentration, our modeling would predict PFOA groundwater concentrations in this scenario from essentially zero to 0.17 ppt. The observed concentration of 1.5-3 ppt are 10-20 times higher than the upper range of the modeled values, which is a similar margin of underestimation observed for the soil media. This again suggests that the presence of PFOA precursors may be resulting in higher than expected levels of PFOA in the field study. Compared to PFOA, which is detected in shallow groundwater immediately after the application of biosolids, PFOS does not become detectable in groundwater until a year after the biosolids were land applied. This also supports the findings of the modeling that PFOS takes more time to impact groundwater than PFOA. The observed PFOS concentration of 0.8 ppt is close to the higher range of the estimated concentration based on our modeling (0.4 ppt). The variability of PFOA and PFOS concentrations in the tile drainage water is likely a function of many factors, including the amount of rainfall in each rain event where tile drainage water was sampled.

The study found that PFOA and PFOS were not detectable in grains harvested from either the experimental or control plot in this study. This finding is in alignment with expectations based on the low soil concentration of PFOA and PFOS in this study and the low observed uptake factors from soils into the grains of plants including wheat.

Though this study has many differences from the scenarios modeled in this risk assessment, it is encouraging that the overall trends in transport behavior between PFOA and PFOS in the agricultural system are similar between the study and the modeled estimates in this risk assessment. The observed soil and groundwater concentrations in this field study are also within the ballpark of the expected values based on extrapolation from this assessment's modeling of a single land application to a field (the reclamation scenario). Finally, this study found that fields with a single low PFOA and PFOS concentration biosolids application have measurably higher PFOA and PFOS soil concentrations than those with no history of biosolids application.

6.2 Biosolids Investigations in Decatur, Alabama

From 1990 to 2008, the Decatur Utilities Dry Creek WWTP in Decatur, Alabama treated wastewater effluent from more than one local industry producing or using PFAS, including PFOA and PFOS. Between 1995 and 2008, the utility supplied over 34,000 DMT of contaminated biosolids to local farmers on approximately 2,000 hectares of fields across three counties (Lindstrom et al., 2011). The 3M company, which was the main producer of PFAS at this site, conducted a study that measured PFAS in various matrices – WWTP effluent, biosolids, input water to the drinking water treatment plant, finished drinking water, leachate from the local MSW landfill, drinking water reservoir (where applicable), and surface water from a small pond – at this city and five others from 1999 to 2001. This study was called the "Multi-City study." The study included four cities with PFAS-related industry (Decatur, AL; Mobile, AL; Columbus, GA; and Pensacola, FL) and two cities without known PFAS-related industry (Cleveland, TN; Port St. Lucia, FL). The results of the Multi-City study show that PFOS concentrations in sludge ranged from not-quantified (detection at a concentration between the detection limit and quantification limit) to 3,120 ppb for PFOS and non-detect to 244 ppb for PFOA. For both PFOA and PFOS, the highest sludge concentrations were found in Decatur (3M, 2001).

Understanding the Multi-City sampling results from the Decatur site is complicated because PFAS were released directly from PFAS industrial facilities, wastewater effluent into the Tennessee River, landfill leachate at regional landfills accepting industrially-impacted waste, and sewage sludge from the local WWTP and from New York City. It is not possible to fully disaggregate impacts from each of these secondary sources in the overall contamination setting at Decatur, especially because the report did not provide specific sampling locations. However, 3M's sampling found that a small waterbody (it is not stated where this waterbody was located with respect to biosolids fields or other release points) had 108 to 114 ng/L PFOS and 57 to 63 ng/L PFOA. Though 3M did not detect PFOA or PFOS in Decatur's drinking water, subsequent analysis by the drinking water utility in 2005 and 2006 found between 30 and 155 ng/L PFOA in finished drinking water.

The 3M Multi-City Study did not include sampling of environmental conditions at any of the sewage sludge land application sites, but EPA researchers investigated water contamination at various land application sites used by the Decatur WWTP (Lindstrom et al., 2011). These researchers collected 51 different water samples, including drinking water wells (n = 6), wells used for other purposes (livestock, watering gardens, washing, n = 13), and surface water (ponds and streams, n = 32). These samples were collected from 21 separate farms that had received contaminated biosolids. In most cases, the water sources (wells or surface water) were either on or within 500 meters of a biosolid applied field. Farms ranged in size from 9 to 308 hectares, with a total area of more than 2000 hectares receiving WWTP biosolids for as long as 12 years. In the well samples, PFOA was detected in four well samples at concentrations ranging from 149 to 6,410 ng/L and PFOS was detected in three samples, with concentrations ranging from 12 to 151 ng/L (the limit of quantification in this study was 10 ng/L for water samples). In surface water samples, PFOA was detected in 24 samples with concentrations ranging from 13.6 to 11,000 ng/L, and PFOS was detected in 12 samples with concentrations ranging from 11.6 to 83.9 ng/L. The size of these ponds was not reported.

Additional results from these Decatur land application sites are published in Washington et al. (2010), which reported PFAS concentrations in soils. These researchers found that PFOA was present in all samples at concentrations ranging from 3 to 317 ng/g (equivalent to 3,000 to 317,000 ppt) and PFOS was present in all but one sample, with concentrations ranging from 1.78 to 325 ng/g (1,780 to 325,000 ppt). The EPA authors of these studies note that there are many unknowns about the PFOA and PFOS content of applied biosolids at each site and the time that elapsed since application; they highlight that the sewage sludge data available is from a period with anomalously high PFOA content in sludge from 2002 to 2006, and it is not known what the PFOA and PFOS content was in the biosolids that were applied to each site. It is also not known what types and concentrations of PFOA and PFOS precursors were present in the sludge that was applied to the sites. Additional data related to these study sites are also published in Yoo et al. 2009, 2010, and 2011.

In 2009, the USDA sampled blood, tissue, and milk from animals that had grazed on fields that had received Decatur WWTP biosolids. The results of this sampling were reported in a CDC ATSDR Health Consultation memo (CDC, 2013). Researchers sampled blood and tissue from 7 cows that had grazed on "high" application fields and 2 cows that had grazed on "minimally" applied fields. At the time, USDA's "minimum proficiency level" for PFOA and PFOS in these blood and tissue samples were 20 ppb (20,000 ppt); results below this level were considered "non detections." They did not detect PFOA or PFOS in these cow tissue or blood samples. The FDA sampled milk from a single dairy cow and milk from a bulk milk tank that was used by regional dairy farms. The single milk sample from the cow did not result in a detection of PFOA or PFOS, but the bulk milk tank had 160 ppt PFOS and no detection of PFOA.

Though the various studies of PFOA and PFOS impacts at and around the Decatur biosolids land application sites do not include all the data necessary to compare this assessment's modeled results to "real life" setting, there are many trends in the Decatur studies that are also seen in the modeled results. First, when PFOA and PFOS-contaminated biosolids were land-applied to fields, these studies show impacted soils, surface waters, groundwater, and dairy cows. These results confirm our modeling that PFOA is more mobile in water than PFOS, causing more widespread impacts to groundwater and surface water. These data also show that while PFOS does migrate to surface water and groundwater, it is more strongly sorbed to soils. Additionally, these data show that PFOS is more likely to be detected in milk than PFOA, which aligns with our higher uptake factors for PFOS than PFOA in dairy cow scenarios.

In this assessment's models, which are assuming PFOA and PFOS have a concentration of 1 ppb in biosolids, groundwater concentrations for PFOA range up to 4.3 ng/L and for PFOS range up to 2 ng/L at pasture farms. Though the exact starting concentrations of PFOA and PFOS in the Decatur sewage sludge that was land-applied at each site is unknown, one can assume that the concentrations of PFOA and PFOS in the sewage sludge applied at these sites were the same as the concentrations reported in 3M's Multi-City study (3,120 ppb PFOS and 244 ppb PFOA). Assuming a linear relationship between PFOA and PFOS concentration in biosolids and their corresponding concentrations in groundwater and that all other biosolids application settings stay constant, this assessment's model would predict ~1,050 ng/L groundwater concentrations for PFOA and ~6,240 ng/L groundwater concentration for PFOS. This PFOA concentration is within the range of observed values in Decatur for groundwater in wells near the land application sites, but the predicted PFOS concentration is higher than the maximum measured PFOS concentration of 151 ng/L. Similarly extrapolating our modeled surface water samples to assume starting conditions of 3,120 ppb PFOS and 244 ppb PFOA gives a modeled value of ~ 2,440 ng/L PFOA and ~400-26,500 ng/L PFOS, depending on the climate and K_{oc} scenario. These surface water modeled results are within the range of observed values for PFOA, but higher than the observed values for PFOS. Given the significant uncertainties around the actual application rate, timing, and PFOA and PFOS concentrations of Decatur biosolids, the farming practices at the farms with the sampled cows, and the size and location of the surface water bodies, modeled and observed values are within a reasonable range.

This study also included samples at a background field that had not received any biosolids. The background field was sampled in 2007 and 2009. In 2007, PFOA and PFOS were not detectable in the top 10 inches of soil. In 2009, PFOA and PFOS were detected at very low levels in the top 10 inches of soil (less than 1 ppt for PFOA and 1 to 2 ppt for PFOS), and slightly higher levels in the deeper soils collected between 38 and 53 inches in depth (PFOA ranging from 64 to 226 ppt and PFOS ranging from ND to 248 ppt). The background results in the top ten inches of soil are consistently below the modeled PFOA and PFOS soil concentrations observed after land applying low concentration (1 ppb) biosolids for 40 years. However, the deeper PFOA and PFOS soil concentrations are slightly higher than modeled in this assessment's land application scenarios. This might reflect the fact that the models assume zero other sources of PFAS to the field, including zero impacts of atmospheric deposition. This site was near a local PFAS industry that may have led to localized atmospheric deposition in soils. Soil studies at remote locations around the globe show that PFOA and PFOS loading in the atmosphere has resulted in small amounts of atmospheric deposition to soils, especially during the time window when PFOA, PFOS, and their precursors were actively being manufactured in large quantities (Rankin et al., 2016). Long-term deposition of PFOA and PFOS at this background site, along with biotic and abiotic mixing of the soil profile, may have led to a build-up of PFOA and PFOS in soil 30 to 50 inches below the surface. PFOA and PFOS at this depth may have less ability to be taken up into grasses or other plants that are used to grow livestock. Similarly, only the top layer of soil, which has low PFOA and PFOS concentrations in this background site, would be relevant for livestock ingestion of soil.

6.3 Biosolids Investigations in Wixom, Michigan

In 2018, Michigan discovered that the Wixon WWTP had been receiving PFAS waste from a local auto supplier conducting chrome plating; biosolids sampled that year were found to have 2,150 ppb PFOS (MPART 2023). PFOA concentrations in the biosolids were much lower, between 1 and 5 ppb (MI EGLE, 2021c). Michigan selected six historic biosolids land application sites used by this WWTP, where they sampled drinking water for humans and livestock, soil surface water, crops, and beef (MI EGLE 2021c). Three of the sampled sites are owned by the same farmer; these sites are fields ranging from 20 to 35 acres. Each site received annual biosolids applications totaling from 184 to 521 DMT over 5 years. Another site from this farmer is 120 acres and received a total of 490 DMT over five years. The last two sites are owned by a second farmer, are 13 to 24 acres, and received from 188 to 242 DMT over 4 or 5 years. The soils at all sites are loamy sand and glacial till; soil borings show interspersed layers of clay.

Soil concentrations at these sites ranged from 2.48 to 96.7 ppb PFOS (2,480-96,700 ppt) and below detection to 1.53 ppb PFOA (detection limit from 800-900 ppt; highest observed concentrations equate to 1,530 ppt). Total organic content of the soils ranged from 7,800 to 12,000 mg/kg. Surface water samples included perched water on the field, water from nearby ponds, water from nearby creeks, and one tile drain sample. Surface water samples ranged from below detection to 533 ppb PFOS (detection limit ~1.5 ppt; up to 533,000 ppt) and below detection to 64.4 ppb PFOA (detection limit also ~1.5 ppt; up to 64,400 ppt). The tile drain sample had a PFOA concentration of 5.98 ppb PFOA and 17.6 ppb PFOS (5,980 and 17,600 ppt). Groundwater monitoring wells were installed and sampled; pre-existing livestock and home drinking water wells were also sampled. Groundwater wells all showed non-detectable levels of PFOA and PFOS (less than 2 ppt). The report authors note that all groundwater wells are screened below a confining clay layer. In a separate advisory, Michigan PFAS Action Response Team

(MPART) shared that beef (roasts and steaks) from one of the sampled farms had between 0.98 and 2.48 ppb PFOS (980-2,480 ppt) (MPART, 2023). There is no publicly available information on the farming practices at this farm, including if feed was sourced from the farm or what the source of drinking water was for the animals.

All except one of the fields in this investigation are smaller than the 80-acre field modeled in this assessment. Like the investigations in Decatur, AL, there are significant uncertainties around the concentration of PFOA and PFOS in the biosolids that were applied at each site. The biosolids application rate is within the range of DMT/hectare modeled in the pasture and reclamation scenarios. These sites accepted biosolids for 5 years; our pasture model assumes annual applications every year for 40 years and our reclamation model assumes a single application.

The modeled soil concentrations in this assessment range from 3 to 790 ppt for PFOA and 21 to 1100 ppt for PFOS in the pasture farm scenario. These sites have soil concentrations ranging from 2 to 10 times the high-end modeled PFOS concentration (2,480 to 96,700 ppt) and mostly within the modeled range for PFOA (less than 900 ppt to 1,530 ppt). Given that our modeled PFOS scenario is for fields more than twice the size of the sampled fields, for application timeframes that amount to 10 times the length of application at these fields, but at concentrations likely 1/2000 of the concentrations in this setting, the soil results in this setting are within the ballpark of what would be expected using our models. The same ballpark agreement is true for PFOA in soils, which was likely applied at concentrations 1 to 5 times the modeled values. Our pasture model scenario found surface water concentrations range from 0.69 to 10 ppt for PFOA and 0.13 to 8.5 ppt for PFOS. In this site, surface water samples ranged from below detection to 64,400 ppt PFOA and from below detection to 533,000 ppt for PFOS. The higher range of these results are higher than expected for PFOA and may reflect higher PFOA concentrations in the applied biosolids than is estimated from the modern-day sample included in the report. The higher end of the PFOS results is also slightly higher than would be expected if the starting concentration of biosolids were ~2,000 times what was modeled, though they are within one or two orders of magnitude. The beef tissue PFOS results that were reported as being associated with grazing on these sites (980 to 2,480 ppt) are 20 times lower than the modeled results on the low end and 250 times lower than the modeled results on the high end. Again, given the potential that biosolids in this setting were 2000 times the modeled results, there are significant differences in the sizes of fields and application rates of biosolids, and there is no information available on the livestock exposure pathways at this farm (*e.g.*, feed, water, soil), the observed results are within the ballpark of what would be expected via our models.

6.4 Biosolids Investigations at Various Farms in Maine

There have been several farms in Maine with PFOA and PFOS impacts from land applying contaminated biosolids to fields later used for growing crops, growing feed for animals, or grazing animals. Though investigations at these farms have sampled milk, hay, crops for human consumption, soil, surface water and groundwater, the specific results for each impacted site have not yet been published in a journal article or public report. Therefore, these sites cannot be used to compare against our modeling exercises.

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APPENDIX A. SUMMARY OF PFOA AND PFOS OCCURRENCE IN BIOSOLIDS IN THE US

This appendix presents concentration data compiled from published peer-reviewed literature and state reports that were available as of January 2024. **Table A-1** presents occurrence data for PFOA and **Table A-2** provides occurrence data for PFOS. **Table A-3** highlights recent studies of PFOA and PFOS precursor occurrence.

A.1 Occurrence of PFOA and PFOS

Table A-1. PFOA Occurrence in Biosolids in the US

Reference	Geographic Area	Years Sampled	Sample Type	PFOA Concentration (Dry Weight Basis): Range (Min-Max), Mean, and/or Median	Method Used	Notes
3M Environmental Laboratory, 2001	US (Multiple states)	1999-2001	WWTP Biosolids	Range: <17 ppb (4 WWTPs) ≤244 ppb (Decatur Utilities Plant)	Modified	Sampled 6 test cities, including Decatur, AL (3M Multi-City Study)
Higgins et al., 2005	US (Multiple states)	1998-2004	WWTP Biosolids	Range: n.d.-29.4 ppb	Modified	Digested sludge samples from 8 WWTPs and primary settled solids from 3 WWTPs (9 WWTPs in total)
Schultz et al., 2006	US (Pacific Northwest Region)	2004	WWTP Biosolids	Range: Digested sludge: <3 ppb	Modified	Analyzed wastewater and sludge samples throughout the treatment process
Sinclair and Kannan, 2006	US (New York)	2005	WWTP Biosolids	Range: Plant A: 69-241 ppb Plant B: 18-89 ppb Mean: Plant A: 144 ppb Plant B: 70 ppb Median: Plant A: 134 ppb Plant B: 80 ppb	Modified	Sampled wastewater at 6 WWTPs, two of which were also sampled for biosolids (five times each)
Loganathan et al., 2007	US (Kentucky and Georgia)	2005	WWTP Biosolids	Range: Plant A: 8.3-219 ppb Plant B: 7-130 ppb	Modified	Sampled two WWTPs: rural (Plant A, Kentucky) and urban (Plant B, Georgia)

Reference	Geographic Area	Years Sampled	Sample Type	PFOA Concentration (Dry Weight Basis): Range (Min-Max), Mean, and/or Median	Method Used	Notes
Yoo et al., 2009	US (Alabama and New York)	2007	WWTP Sewage Sludge/ Biosolids	Mean: Decatur WWTP: Sample A: 50.3±4.7 ppb Sample B: 128±8.3 ppb New York City WWTPs: Sample A: 8.7±0.7 ppb Sample B: 8.4±5 ppb Sample C: 20±3.9 ppb	Modified Isotopic-Dilution Method with LC-MS/MS	Conducted a method development study for measuring PFAS, using sludge samples from a WWTP in Decatur, AL; this method was then used to assess PFAS in a NIST sludge sample and sludge samples from New York City WWTPs
Washington et al., 2010	US (Alabama)	2007 and 2009	Land-applied Biosolids	Range: 2009: ≤320 ppb	Modified	Conducted two sampling surveys (2007 and 2009)
Lindstrom et al., 2011	US (Alabama)	2009	Well and Surface Water near Land-applied Biosolids Sites	Range: Well Water: <LOQ-6410 ppt Surface Water: <LOQ-11,000 ppt	Modified	Sampled well and surface water sites near historical biosolids land application in Decatur, AL Well and Surface Water: PFOA Detection Rate = 57%
Sepulvado et al., 2011	US (Illinois)	2004-2007	Land-applied Biosolids	Range: 8-68 ppb	Modified	Compiled 6 composite samples PFOA Detection Rate = 100%
Venkatesan and Halden, 2013	US (Multiple states)	2001	WWTP Biosolids	Range: 12-70 ppb Mean: 34±22 ppb	Modified EPA Method 1694	Compiled 5 composite samples from 110 archived biosolids samples from the US EPA 2001 NSSS (94 POTWs) PFOA Detection Rate = 100%
Armstrong et al., 2016	US (Mid-Atlantic Region)	2005-2013	WWTP Biosolids	Mean: 23.5 ppb Median: 2.5 ppb	Modified	Performed temporal trend study (multiple samples collected between 2005 and 2013 from 1 urban WWTP)
Lazcano et al., 2020	US (Multiple states)	2014, 2016, 2018	Biosolids-based Products	Range: Biosolids-based products: 1.4-26 ppb	Modified	Analyzed multiple types of biosolids-based and non-biosolids organic products
Pepper et al., 2021	US (Arizona)	2020	WWTP Biosolids and Land-applied Biosolids	Range: Biosolids: ≤1.2 ppb	Modified EPA Method 537.1	Collected samples in 2020 from a WWTP in Arizona and field sites where Class B biosolids were land applied from 1984-2019
Helmer et al., 2022	US (Michigan)	2018-2020	WWTP Biosolids	For 1 of 11 samples, PFOA slightly dominant (207 ppb, calculated from total concentration and percent composition)	Modified EPA Method 537.1	Analyzed 11 samples from 6 industrially impacted WWTPs
Johnson, 2022	US (Western Region)	2015	Land-applied Biosolids	Mean: 0.44* ppb	Modified	Collected 2 biosolids samples

Reference	Geographic Area	Years Sampled	Sample Type	PFOA Concentration (Dry Weight Basis): Range (Min-Max), Mean, and/or Median	Method Used	Notes
Schaefer et al., 2022	US (Multiple states)	2020	WWTP Biosolids and Column Mesocosm Leaching Experiments	Range: 0.8-8.12 ppb	Modified	Sampled 7 WWTPs with a variety of treatment processes in urban areas, receiving both industrial and domestic sources, as well as performed column mesocosm leaching experiments; found that PFAS precursors accounted for over 75% of total PFAS
Thompson et al., 2023a	US (Florida)	2021	WWTP Sewage Sludge/ Biosolids	Range: Sludge (Before Treatment): 1.7-21 ppb Biosolids (After Treatment): 1.1-7.7 ppb	Modified	Interviewed 39 facilities in Florida to learn treatment processes from 2019-2021; Then, in 2021, collected 16 samples (before and after treatment) from 8 facilities representing the four most common treatment processes; studied 92 PFAS analytes, including precursors
Thompson et al., 2023b	US (Florida)	Sludge: 2021 Toilet Paper: 2021-2022	WWTP Sewage Sludge/ Biosolids and Toilet Paper	Range: Sludge: 1.7-21 ppb Toilet Paper: <LOD-0.2 ppb	Modified	Focused on studying diPAPs in sludge (Florida, US) and toilet paper samples (US and other countries)
Link et al., 2024	US (Michigan)	2018-2022	WWTP Biosolids	Range: <96 ppb Mean: 4.8±11 ppb	Modified EPA Method 537.1	Sampled 190 WWTPs, representing both industrial and domestic sources PFOA Detection Rate = 63%
USGS/NH DES: Phase 1: Santangelo et al., 2022; Phase 2: Tokranov et al., 2023; Phase 3: Santangelo et al., 2023	US (New Hampshire)	2021-2022	Soils, Land-applied Biosolids, Solid/Water Partitioning, and Groundwater Leaching	Range: Finished biosolids (collected from facilities in 2021 as part of Phase 2): 0.67*-7.5 ppb	Eurofins LC-MS/MS and Isotope Dilution	Three-phase study of soils, land-applied biosolids, solid/water partitioning, and groundwater leaching
San Francisco Estuary Institute (SFEI): Phase 1: Mendez et al., 2021	US (California)	2020	WWTP Biosolids	Range: n.d.-15 ppb Mean: 3 ppb Median: 1 ppb	SGS AXYS Method MLA-110	PFAS Study of Bay Area WWTPs: Phase 1
MPCA, 2008	US (Minnesota)	2007-2008	WWTP Sewage Sludge	Range: 2007: <0.191**-54.6 ppb 2008: <0.748-35.4 ppb	Modified	Monitored PFAS at WWTPs in 2007 and 2008

Reference	Geographic Area	Years Sampled	Sample Type	PFOA Concentration (Dry Weight Basis): Range (Min-Max), Mean, and/or Median	Method Used	Notes
CT DEEP: Weston & Sampson, 2023	US (Connecticut)	2021-2022	WWTP Biosolids	Range: Sludge (liquid): 0-51 ppt Sludge (solid): 0-13 ppb Mean: Sludge (liquid): 13 ppt Sludge (solid): 1 ppb Median: Sludge (liquid): 8.6 ppt Sludge (solid): 0 ppb	Modified	Study of PFAS in WWTPs PFOA Detection Rate: Sludge (liquid): 90% Sludge (solid): 23%
VT DEC: Weston & Sampson, 2020	US (Vermont)	2018-2019	WWTP Biosolids	Range: Sludge (liquid): Average sum of 5 VT DEC regulated PFAS (PFHxS, PFHpA, PFOA, PFOS, and PFNA) across WWTPs: <80 ppt, except one facility at 505 ppt Sludge (solid): Average sum of 5 VT DEC regulated PFAS (PFHxS, PFHpA, PFOA, PFOS, and PFNA) across WWTPs: 5-50 ppb, except one facility at 85 ppb	Modified EPA Method 537.1	Study of PFAS in landfill leachate and WWTPs Collected 75 sludge samples: Report summarized results as sum of 5 VT DEC regulated PFAS (PFHxS, PFHpA, PFOA, PFOS, and PFNA)
VT DEC: Weston & Sampson, 2022	US (Vermont)	2021	PFAS Sources to WWTPs	PFOA commonly detected in sources (residential, commercial, and industrial inputs)	Modified EPA Method 537.1	Study of PFAS sources to WWTPs
Maine DEP: Brown and Caldwell, 2023	US (Maine)	2019-2022	WWTP Biosolids	Range: 2019: n.d.-46 ppb 2020: 0.6-63 ppb 2021: 0.3-25 ppb 2022: 0.8-38.9 ppb Mean: 2019: 9.4 ppb 2020: 8.2 ppb 2021: 5.3 ppb 2022: 6.6 ppb	Modified EPA Method 537.1	Based on biosolids data in Maine's Environmental and Geographic Analysis Database collected from 2019-2022

LOQ = Limit of Quantification
LOD = Limit of Detection
n.d. = non-detect
* below reporting limit or limit of detection
** estimated value based on quality assurance review

Table A-2. PFOS Occurrence in Biosolids in the US

Reference	Geographic Area	Years Sampled	Sample Type	PFOS Concentration (Dry Weight Basis): Range (Min-Max), Mean, and/or Median	Method Used	Notes
3M Environmental Laboratory, 2001	US (Multiple states)	1999-2001	WWTP Biosolids	Range: 58-159 ppb (4 WWTPs) ≤3120 ppb (Decatur Utilities Plant)	Modified	Sampled 6 test cities, including Decatur, AL (3M Multi-City Study)
Higgins et al., 2005	US (Multiple states)	1998-2004	WWTP Biosolids	Range: 14.4-2610 ppb Mean (Post-2002): 124 ppb (n=8)	Modified	Digested sludge samples from 8 WWTPs and primary settled solids from 3 WWTPs (9 WWTPs in total)
Schultz et al., 2006	US (Pacific Northwest Region)	2004	WWTP Biosolids	Range: Digested sludge: 81-160 ppb	Modified	Analyzed wastewater and sludge samples throughout the treatment process
Sinclair and Kannan, 2006	US (New York)	2005	WWTP Biosolids	Range: Plant A: 26-65 ppb Plant B: <10-34 ppb Mean: Plant A: 37 ppb Plant B: 25 ppb Median: Plant A: 28 ppb Plant B: 32 ppb	Modified	Sampled wastewater at 6 WWTPs, two of which were also sampled for biosolids (five times each)
Loganathan et al., 2007	US (Kentucky and Georgia)	2005	WWTP Biosolids	Range: Plant A: 8.2-990 ppb Plant B: <2.5-77 ppb	Modified	Sampled two WWTPs: rural (Plant A, Kentucky) and urban (Plant B, Georgia)
Yoo et al., 2009	US (Alabama and New York)	2007	WWTP Sewage Sludge/ Biosolids	Mean: Decatur WWTP: Sample A: 346.3±44.4 ppb Sample B: 417.9±57.2 ppb New York City WWTPs: Sample A: 76.8±27.8 ppb Sample B: 61.1±17.1 ppb Sample C: 32.2±0.7 ppb	Modified Isotopic-Dilution Method with LC-MS/MS	Conducted a method development study for measuring PFAS, using sludge samples from a WWTP in Decatur, AL; this method was then used to assess PFAS in a NIST sludge sample and sludge samples from New York City WWTPs
Washington et al., 2010	US (Alabama)	2007 and 2009	Land-applied Biosolids	Range: 2009: ≤410 ppb	Modified	Conducted two sampling surveys (2007 and 2009)
Lindstrom et al., 2011	US (Alabama)	2009	Well and Surface Water near Land-applied Biosolids Sites	Range: Well Water: <LOQ-151 ppt Surface Water: <LOQ-83.9 ppt	Modified	Sampled well and surface water sites near historical biosolids land application in Decatur, AL Well and Surface Water: PFOS Detection Rate = 29%

Reference	Geographic Area	Years Sampled	Sample Type	PFOS Concentration (Dry Weight Basis): Range (Min-Max), Mean, and/or Median	Method Used	Notes
Sepulvado et al., 2011	US (Illinois)	2004-2007	Land-applied Biosolids	Range: 80-219 ppb Mean: 144±57 ppb	Modified	Compiled 6 composite samples PFOS Detection Rate = 100%
Venkatesan and Halden, 2013	US (Multiple states)	2001	WWTP Biosolids	Range: 308-618 ppb Mean: 403±127 ppb	Modified EPA Method 1694	Compiled 5 composite samples from 110 archived biosolids samples from the US EPA 2001 NSSS (94 POTWs) PFOS Detection Rate = 100%
Armstrong et al., 2016	US (Mid-Atlantic Region)	2005-2013	WWTP Biosolids	Mean: 22.5 ppb Median: 19.3 ppb	Modified	Performed temporal trend study (multiple samples collected between 2005 and 2013 from 1 urban WWTP)
Lazcano et al., 2020	US (Multiple states)	2014, 2016, 2018	Biosolids-based Products	Range: Biosolids-based products: 2.6-88.5 ppb	Modified	Analyzed multiple types of biosolids-based and non-biosolids organic products
Pepper et al., 2021	US (Arizona)	2020	WWTP Biosolids and Land-applied Biosolids	Range: Biosolids: 14-36 ppb	Modified EPA Method 537.1	Collected samples in 2020 from a WWTP in Arizona and field sites where Class B biosolids were land applied from 1984-2019
Helmer et al., 2022	US (Michigan)	2018-2020	WWTP Biosolids	Range: 4-6500 ppb For 8 of 11 samples, PFOS dominant	Modified EPA Method 537.1	Analyzed 11 samples from 6 industrially impacted WWTPs; PFOS was the dominant type of PFAS measured in 8 of the 11 biosolids samples (~73%)
Johnson, 2022	US (Western Region)	2015	Land-applied Biosolids	Mean: 12 ppb	Modified	Collected 2 biosolids samples
Schaefer et al., 2022	US (Multiple states)	2020	WWTP Biosolids and Column Mesocosm Leaching Experiments	Range: 0.386-150 ppb	Modified	Sampled 7 WWTPs with a variety of treatment processes in urban areas, receiving both industrial and domestic sources, as well as performed column mesocosm leaching experiments; found that PFAS precursors accounted for over 75% of total PFAS
Thompson et al., 2023a	US (Florida)	2021	WWTP Sewage Sludge/ Biosolids	Range: Sludge (Before Treatment): 4-41 ppb Biosolids (After Treatment): 1.4-19 ppb	Modified	Interviewed 39 facilities in Florida to learn treatment processes from 2019-2021; Then, in 2021, collected 16 samples (before and after treatment) from 8 facilities representing the four most common treatment processes; studied 92 PFAS analytes, including precursors
Thompson et al., 2023b	US (Florida)	Sludge: 2021 Toilet Paper: 2021-2022	WWTP Sewage Sludge/ Biosolids and Toilet Paper	Range: Sludge: 4-41 ppb	Modified	Focused on studying diPAPs in sludge (Florida, US) and toilet paper samples (US and other countries)
Link et al., 2024	US (Michigan)	2018-2022	WWTP Biosolids	Range: <2150 ppb Mean: 40±179 ppb	Modified EPA Method 537.1	Sampled 190 WWTPs PFOS Detection Rate = 95%

Reference	Geographic Area	Years Sampled	Sample Type	PFOS Concentration (Dry Weight Basis): Range (Min-Max), Mean, and/or Median	Method Used	Notes
MI EGLE, 2021	US (Michigan)	2018-2021	WWTP Biosolids	Industrially Impacted: Range: 360-6500 ppb Not Industrially Impacted: Mean: 18 ppb Median: 11 ppb	Modified EPA Method 537.1	State PFAS Survey – Interim Strategy: Surveyed 42 WWTPs; Industrially impacted: 6 WWTPs
MI EGLE, 2022	US (Michigan)	2017/ 2018, 2021	WWTP Biosolids	Industrially Impacted: Range: 2017/2018: 160-2150 ppb 2021: 33-180 ppb	Modified EPA Method 537.1	Update to State PFAS Survey – Interim Strategy
USGS/NH DES: Phase 1: Santangelo et al., 2022; Phase 2: Tokranov et al., 2023; Phase 3: Santangelo et al., 2023	US (New Hampshire)	2021-2022	Soils, Land-applied Biosolids, Solid/Water Partitioning, and Groundwater Leaching	Range: Finished biosolids (collected from facilities in 2021 as part of Phase 2): 2.2-7.9 ppb	Eurofins LC-MS/MS and Isotope Dilution	Three-phase PFAS study of soils, land-applied biosolids, solid/water partitioning, and groundwater leaching
San Francisco Estuary Institute (SFEI): Phase 1: Mendez et al., 2021	US (California)	2020	WWTP Biosolids	Range: n.d.-49 ppb Mean: 14 ppb Median: 13 ppb	SGS AXYS Method MLA-110	PFAS Study of Bay Area WWTPs: Phase 1
MPCA, 2008	US (Minnesota)	2007-2008	WWTP Sewage Sludge	Range: 2007: <0.382**-861 ppb 2008: 4.15**-442 ppb	Modified	Monitored PFAS at WWTPs in 2007 and 2008
CT DEEP: Weston & Sampson, 2023	US (Connecticut)	2021-2022	WWTP Biosolids	Range: Sludge (liquid): 0-21 ppt Sludge (solid): 0-43 ppb Mean: Sludge (liquid): 7 ppt Sludge (solid): 12.4 ppb Median: Sludge (liquid): 4.9 ppt Sludge (solid): 10 ppb	Modified	Study of PFAS in WWTPs PFOS Detection Rate: Sludge (liquid): 70% Sludge (solid): 85%

Reference	Geographic Area	Years Sampled	Sample Type	PFOS Concentration (Dry Weight Basis): Range (Min-Max), Mean, and/or Median	Method Used	Notes
VT DEC: Weston & Sampson, 2020	US (Vermont)	2018-2019	WWTP Biosolids	Range: Sludge (liquid): Average sum of 5 VT DEC regulated PFAS (PFHxS, PFHpA, PFOA, PFOS, and PFNA) across WWTPs: <80 ppt, except one facility at 505 ppt Sludge (solid): Average sum of 5 VT DEC regulated PFAS (PFHxS, PFHpA, PFOA, PFOS, and PFNA) across WWTPs: 5-50 ppb, except one facility at 85 ppb	Modified EPA Method 537.1	Study of PFAS in landfill leachate and WWTPs Collected 75 sludge samples: Report summarized results as sum of 5 VT DEC regulated PFAS (PFHxS, PFHpA, PFOA, PFOS, and PFNA)
VT DEC: Weston & Sampson, 2022	US (Vermont)	2021	PFAS Sources to WWTPs	PFOS commonly detected in sources (residential, commercial, and industrial inputs)	Modified EPA Method 537.1	Study of PFAS sources to WWTPs
Maine DEP: Brown and Caldwell, 2023	US (Maine)	2019-2022	WWTP Biosolids	Range: 2019: 2.2-120 ppb 2020: 2.5-51.9 ppb 2021: 2.1-111 ppb 2022: 1.2-66 ppb Mean: 2019: 27.2 ppb 2020: 16.6 ppb 2021: 22.7 ppb 2022: 19.3 ppb	Modified EPA Method 537.1	Based on biosolids data in Maine's Environmental and Geographic Analysis Database collected from 2019-2022

LOQ = Limit of Quantification
LOD = Limit of Detection
n.d. = non-detect
* below reporting limit or limit of detection
** estimated value based on quality assurance review

A.2 Occurrence of PFOA and PFOS Precursors

Table A-3. Recent Examples of PFOA and PFOS Precursor Occurrence in Biosolids in the US

Reference	Geographic Area	Years Sampled	Sample Type	PFOA Precursor: 8:2 diPAP Concentration (Dry Weight Basis): Range (Min-Max)	PFOS Precursor: NEtFOSAA Concentration (Dry Weight Basis): Range (Min-Max)	Method Used	Notes
Schaefer et al., 2022	US (Multiple states)	2020	WWTP Biosolids and Column Mesocosm Leaching Experiments	Range: 13.5-347 ppb	Range: 0.297-18 ppb	Modified	Sampled 7 WWTPs with a variety of treatment processes in urban areas, receiving both industrial and domestic sources, as well as performed column mesocosm leaching experiments; found that PFAS precursors accounted for over 75% of total PFAS
Thompson et al., 2023a	US (Florida)	2021	WWTP Sewage Sludge/ Biosolids	Range: Sludge (Before Treatment): 21-300 ppb Biosolids (After Treatment): 5.9-100 ppb	Range: Sludge (Before Treatment): 0-7.6 ppb Biosolids (After Treatment): 0-3.9 ppb	Modified	Interviewed 39 facilities in Florida to learn treatment processes from 2019-2021; Then, in 2021, collected 16 samples (before and after treatment) from 8 facilities representing the four most common treatment processes; studied 92 PFAS analytes, including precursors
Thompson et al., 2023b	US (Florida)	Sludge: 2021 Toilet Paper: 2021-2022	WWTP Sewage Sludge/ Biosolids and Toilet Paper	Range: Sludge: 21-300 ppb Toilet Paper: <LOD-0.2 ppb	Range: Sludge: 0-7.6 ppb	Modified	Focused on studying diPAPs in sludge (Florida, US) and toilet paper samples (US and other countries)

LOD = Limit of Detection

A.3 References

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APPENDIX B. MODEL INPUTS

This appendix is organized by the component models used in this assessment, as follows:

- B.1 General Inputs (used by multiple models)
- B.2 Land Application Unit Source Model (3MRA LAU Source Module)
- B.3 Surface Disposal Unit Source Model (3MRA SI Module)
- B.4 Groundwater Model (EPACMTP)
- B.5 Surface Water Model (VVWM)
- B.6 Food Chain Calculations
- B.7 Exposure Calculations
- B.8 Risk Calculations.

Within any section, multiple tables may be provided if inputs vary with scenario, chemical, or climate location. Within each table, inputs are listed alphabetically. Note that some values may be rounded for clarity of presentation.

B.1 General Inputs

Table B-1. Chemical-specific Inputs

Parameter	Description & Units	PFOA	PFOS	Reference	Comment
ChemType	Type of chemical (e.g., organic, metal/inorganic, mercury, dioxin-like)	O	O	NA	This parameter is used by the source and food chain models to identify the appropriate algorithms and inputs, as these differ between organics and inorganics. PFOA and PFOS are both organic chemicals.
Da	Diffusivity in air (cm^2/s)	NA	NA	NA	This assessment does not include modeling transport through air
Dw	Diffusion coefficient in water (cm^2/s)	5.52E-06	4.96E-06	US EPA (2016)	
HLC	Henry's law constant [atm-m^3/mol]	NA	NA	NA	Volatilization is not expected under environmental conditions (see Section 2.2.2)
Koc-high	Organic carbon partition coefficient (high end) [mL/g]	1,100	22,000	PFOA: Campos-Pereira et al., 2023; PFOS: Chen et al., 2020	90th percentile from literature search; n = 203 for PFOA, 253 for PFOS; see Appendix C for more details
Koc-low	Organic carbon partition coefficient (low end) [mL/g]	26	250	PFOA: Hubert, M., et al, 2023; PFOS: Johnson et al., 2007	10th percentile from literature search; n = 203 for PFOA, 253 for PFOS; see Appendix C for more details
MW	Molecular weight [g/mol]	414	500	PFOA: HSDB (US NLM, 2010); PFOS: Physprop (SRC, 2016)	

B.2 Regional Location-based Parameters

In addition to general chemical parameters, the assessment modeled three climates—dry, moderate, and wet—represented by specific meteorological stations. These were chosen based on the number of precipitation days per year, not total annual rainfall. The meteorological stations and their general descriptive data are as follows (all from SAMSON—US DOC & US DOE, 1993):

- Dry climate:
  - Location of meteorological station: Boulder, CO
  - WBAN station number: 94018
  - Meteorological station latitude: 40.0167°
  - Long-term average annual air temperature: 10.11 °C
- Moderate Climate:
  - Location of meteorological station: Chicago, IL
  - WBAN station number: 94846
  - Meteorological station latitude: 41.983°
  - Long-term average annual air temperature: 9.69 °C
- Wet Climate:
  - Location of meteorological station: Charleston, SC
  - WBAN station number: 13880
  - Meteorological station latitude: 32.9°
  - Long-term average annual air temperature: 18.18 °C

B.3 LAU Source Model Inputs (3MRA LAU Module)

Chemical-, scenario-, and location-specific inputs are presented in **Tables B-2**, **B-3, and B-4**, respectively. The LAU Source Module has three submodules, the Generic Soil Colum Model (GSCM), which evaluates movement vertically through the soil column; the Local Watershed Model (LWS), which evaluates movement horizontally onto and off the field; and the Particulate Emissions Model (PEM), which accounts for particulate emissions to air. The PEM accounts for losses only; this assessment does not model transport through air. The “Used in” column notes which submodel uses an input (or says “LAU” if the input is general to all submodules).

Inputs for which there is a single value (*i.e.*, they are not specific to a chemical, scenario, or location) are presented in **Table B-5**, grouped by LAU submodel.

Table B-2. Chemical-Specific Inputs to the LAU Source Module

Parameter	Description & Units	Used in	PFOA	PFOS	Reference	Comment
ChemFracNeutral	Fraction of chemical concentration in the neutral species (fraction)	GSCM	NA	NA	NA	Used to adjust properties for chemicals that ionize; not applicable to this assessment.
ChemTemp	Temperature (°C)	GSCM	NA	NA	NA	Temperature at which degradation and volatilization rates are measured; not applicable to this assessment
ksoil	Soil biodegradation rate (1/day)	GSCM	0	0	NA	Based on PFOA/PFOS degradation literature
Sol	Solubility [mg/L]	GSCM	9500	680	US EPA (2017a)	Used to determine if solubility is exceeded in the soil column during model run

Table B-3. Scenario-Specific Inputs to the LAU Module

Parameter	Description & Units	Used in	Crop	Pasture	Reclamation	Reference	Comment
AppDepth	Depth of biosolids incorporation (m)	GSCM	0.2	0.02	0.02	Assumption	For the crop scenario, biosolids are tilled into the soil to a depth of 20 cm at application. For the pasture and reclamation scenarios, the biosolids are not tilled in, but are assumed to be incorporated to a depth of 2 cm by bioturbation. This assumption is consistent with the 2003 Biosolids assessment, US EPA (2003a).
CN_wmu	SCS curve number for field (dimensionless ratio)	LWS	81	71	71	USDA (1986)	Average across hydrologic soil groups and hydrologic conditions for straight row crops (crop scenario) or pasture lands (pasture, reclamation scenarios)
DryApplRate	Application rate of biosolids to the field, dry weight per application (MT DW/ha/appl)	LAU	10	10	50	Crop & pasture: Biosolids Tool (BST; US EPA, 2023a; see Appendix E); Reclamation: Sopper (1993)	Crop & Pasture: median of agronomic rates from probabilistic plant available nitrogen (PAN) calculations conducted for the BST; Reclamation: 5 x agronomic rate
fcult	Number of cultivations per application (count)	PEM	5	1	1	TSDF Fugit. Air (US EPA, 1989b)	Impacts spreading and compacting losses
fd	Frequency of surface disturbance per month on field (1/mo)	PEM	0.21	0.042	0.042	Biosolids 2003 (US EPA, 2003a)	Impacts wind erosion losses
OpLife	Number of years biosolids are applied to the field (years)	LAU	40	40	1	Assumption	Chosen for consistency with 2003 Biosolids assessment (US EPA, 2003a) and 3MRA default value (US EPA, 2003b)
Pwmu	USLE erosion control factor for field (fraction)	LWS	0.5	1	1	Wanielista & Yousef (1993)	a value of 1 means no erosion control practices; these are the 3MRA defaults.
Rappl	Application rate of biosolids to the field, whole weight per year (MT WW/m^2-year)	LAU	0.0025	0.0025	0.0125	Calculated	[DryApplRate x Nappl x 1E-4 ha/m2]/[%solids/100]
zruf	Roughness height of the field (cm)	PEM	1	3.7	3.7	TSDF Fugit. Air (US EPA, 1989b)	Impacts wind erosion losses

Table B-4. Location-Specific Inputs to the LAU Module

Parameter	Description & Units	Used in	Dry	Moderate	Wet	Reference
R	USLE rainfall/erosivity factor (1/year)	LWS	50	155	360	Wischmeier and Smith (1978)
Uw	Mean annual wind speed (m/sec)	PEM	3.783	4.632	3.788	SAMSON (US DOC & US DOE, 1993)

Table B-5. Individual Inputs to the LAU Module

Parameter	Description & Units	Value	Reference	Comment
General LAU Module Inputs				
%solid	Percent solids of biosolids applied to field (mass percent)	48	TNSSS (US EPA, 2009)	Midpoint of range (0.14–94.9%) based on 84 samples
Area_field	Area of the agricultural field (m^2)	323,750	USDA (2014)	80 acres
Nappl	Number of biosolids applications per year (1/year)	1	Assumption	Application is assumed to occur on April 1, at the start of the growing season.
Ss	Silt content of soil (mass %)	42.5	STATSGO (USDA, 1994)	area weighted average for each soil texture within met region – median value
WSpH	Soil pH (pH units)	NA	NA	Used for ionizable chemicals to adjust properties; not applicable to this assessment
GSCM Inputs				
BDwaste	Dry bulk density of biosolids applied to field (g/cm3)	0.7	Gunn et al. (2004)	
foc_biosolids	Fraction organic carbon of biosolids applied to field (fraction)	0.4	Biosolids 2003 (US EPA 2003a)	
foc_soil	Fraction organic carbon for natural soil in the soil column under the field (fraction)	0.0118	STATSGO (USDA, 1994)	Calculated using percent organic matter from STATSGO, based on EPACMTP – median value
fwmu	Fraction of waste in LAU (fraction)	1	Assumption	Indicates that all sewage sludge is applied to field
Ksat	Saturated hydraulic conductivity of soil (cm/h)	0.45	Carsel & Parrish (1988)	based on surface soil textures – median value
WCS	Saturated volumetric water content, porosity for soil (mL/cm^3)	0.43	Carsel & Parrish (1988)	based on surface soil textures – median value
LWS Inputs				
Area_buffer	Area of the buffer between the field and the waterbody (m^2)	5690	Calculated	=length of source x buffer width; length is 569 m, width is 10 m per Part 503 Biosolids rule; ~1.4 acres
C	USLE cover factor (fraction)	0.1	HHRAP (US EPA, 2005)	
CN_buffer	SCS curve number (dimensionless ratio)	69	USDA (1986)	Average across hydrologic soil groups and hydrologic conditions for good pasture and farmsteads
ConVs	Settling velocity of suspended solids in runoff from field (m/day)	5.36	Schroeder (1977)	derived from "mineral sludge" values - median value
DRZ	Root zone depth (cm)	82.7	Dunne & Leopold (1978)	median value
K	USLE soil erodibility factor (kg/m^2)	0.0716	STATSGO (USDA, 1994)	area weighted average for each soil texture within met region – median value
LS	USLE length-slope factor (empirical)	1.5	HHRAP (US EPA, 2005)	Default assessment values from HHRAP
P_buffer	USLE erosion control factor for buffer (fraction)	1	Wanielista & Yousef (1993)	A value of 1 means no erosion control practices. These are the 3MRA defaults.

Parameter	Description & Units	Value	Reference	Comment
SMb	Soil moisture coefficient (vol %)	5.3	Clapp & Hornberger (1978)	based on surface soil textures – median value
SMFC	Soil moisture field capacity (vol %)	22.48	Carsel et al. (1988)	based on average hydrologic soil group for each soil texture – median value
SMWP	Soil moisture wilting point (vol %)	11.48	Carsel et al. (1988)	based on average hydrologic soil group for each soil texture – median value
Theta	Slope of watershed (degrees)	3.66	STATSGO (USDA, 1994)	area weighted average for each soil texture within met region – median value
X	Flow length for local watershed (m)	129	Mills et al. (1985)	calculated from theta and LS using equation in cited reference
PEM Inputs				
asdm	Mode value of the size of soil aggregates in an LAU (mm)	0.5	3MRA (US EPA, 2003b)	3MRA default
effdust	Dust suppression control efficiency (field) (fraction)	0	NA	no regular vehicular activity
Lc	Soil roughness ratio (dimensionless ratio)	2.31E-04	TSDF Fugit. Air (US EPA, 1989b)	
mt	Distance vehicle travels on field (m)	0	NA	no regular vehicular activity
nv	Number of vehicles per day on field (1/day)	0	NA	no regular vehicular activity
nw	Number of wheels on each vehicle (count)	4	NA	no regular vehicular activity
Sw	Silt content of biosolids (mass %)	10	AP-42 (US EPA, 1995)	
veg	Fraction vegetative cover for the field (fraction)	0.8	Assumption	This is the minimum of the assumed 3MRA distribution (which is 0.8 – 1, mean of 0.9, assumed normal). That's based on "best professional judgement, assuming unit is vegetated during operation and after closure."
vw	Vehicle weight (MT)	0	NA	no regular vehicular activity

B.4 Surface Disposal Unit Source Model Inputs (3MRA Surface Impoundment Module)

Table B-6. SDU Inputs

Parameter	Description & Units	Value	Reference	Comment
General Parameters				
Area_SI	Area of the SDU (m^2)	3,400	calculated	=Qwmu/(dwmu * EconLife)
Bio_yield	Biomass yield of the SDU (g/g)	0.6	Tchobanoglous et al. (1979)	Median; generally ranges from 0.4 to 0.8
d_wmu	Depth of the SDU (m)	2	3MRA (US EPA, 2003b)	Median of data from Industrial D Screening Survey
DBGS	Depth of SDU below ground surface (m)	0	EPACMTP (US EPA, 2003c)	
EconLife	Operating life of surface disposal unit (yr)	50	3MRA (US EPA, 2003b)	3MRA default
Q_wmu	Volumetric flow rate into SDU (m^3/s)	4E-06	3MRA (US EPA, 2003b)	Median of data from Industrial D Screening Survey
Waste Parameters				
C_in	Concentration of constituent in SDU influent (mg/L)		Calculated	CTPWasteDry * TSS_in
CBOD	Biological oxygen demand of SDU influent (g/cm3)	8E-3	Tchobanoglous et al. (1979)	Tbl 3-6, typical value for untreated septage
dmeanTSS	Particle diameter of solids in SDU (cm)	0.001	Tchobanoglous et al. (1979)	Default value from the surface impoundment module of 3MRA
kba1	Biologically active solids/total solids ratio in SDU (unitless)	0.4	Tchobanoglous et al. (1979)	Tbl 11-4, typical value for digested sludge
rho_part	Density of solids in SDU (g/cm3)	2.5	Tchobanoglous et al. (1979)	Default value from the surface impoundment module of 3MRA

Parameter	Description & Units	Value	Reference	Comment
SrcPh	pH of SDU influent (pH units)	NA	NA	Used to adjust chemical properties for ionizable chemicals for temp and pH; not applicable to PFOA/PFOS
SrcTemp	Temperature of waste in SDU (°C)	NA	NA	Used to adjust chemical properties for ionizable chemicals for temp and pH; not applicable to PFOA/PFOS
TSS_in	Total suspended solids in SDU influent (g/cm3)	0.1	Tchobanoglous et al. (1979)	Tbl 11-4, typical value for digested sludge
TSS_out	Total suspended solids in SDU effluent (g/cm3)	NA	NA	This assessment assumes that there is no effluent from the surface disposal site
Sediment Layer Properties				
d_setpt	Max fraction of SDU area occupied by sediments (fraction)	0.5	3MRA (US EPA, 2003b)	Median of data from Industrial D Screening Survey
hydc_ssed	Hydraulic conductivity of the SDU sediment layer (m/s)	5E-07	Tchobanoglous et al. (1979)	Median; generally ranges from 1E-9 to 1E-6
k_dec	Digestion rate of sediments in the SDU (1/s)	7E-07	Tchobanoglous et al. (1979)	Median; generally ranges from 4.6E-7 to 8.7E-7
SedAlpha	Soil retention parameter alpha of SDU sediment (1/cm)	0.016	Carsel and Parrish (1988)	Mean for silt soils
SedBeta	Soil retention parameter beta of SDU sediment (unitless)	1.37	Carsel and Parrish (1988)	Mean for silt soils
Liner Properties (used to calculate leachate quantity to pass to EPACMTP)				
d_liner	Thickness of clay liner (m)	0.9144	EPACMTP (US EPA, 2003c)	Default
hydc_liner	Saturated conductivity of clay liner (m/s)	1E-09	EPACMTP (US EPA, 2003c)	Default
Infil_CompLiner	Infiltration rate through composite liner (m/d)	1.4E-06	EPACMTP (US EPA, 2003c)	90th percentile (Table 4.6)
LinerAlpha	Soil retention parameter alpha of the SDU liner (1/cm)	0.008	Carsel and Parrish (1988)	Mean for clay soils
LinerBeta	Soil retention parameter beta of the SDU liner (unitless)	1.09	Carsel and Parrish (1988)	Mean for clay soils
Vadose Zone and Aquifer Properties (used to calculate amount of infiltration to pass to EPACMTP)				
AquSATK	Saturated hydraulic conductivity (m/yr)	1890	EPACMTP (US EPA, 2003c)	National median values; the SDU source model uses these to estimate infiltration rate and does not distinguish location; the GW modeling uses location-specific values.
AquThick	Saturated zone thickness (m)	14.3	EPACMTP (US EPA, 2003c)	
VadAlpha	Soil retention parameter alpha (1/cm)	0.0152	EPACMTP (US EPA, 2003c)	
VadBeta	Soil retention parameter beta (unitless)	1.37	EPACMTP (US EPA, 2003c)	
VadSATK	Saturated hydraulic conductivity of vadose zone soil (cm/h)	0.0089	EPACMTP (US EPA, 2003c)	
VadThick	Thickness of vadose zone (m)	6.1	EPACMTP (US EPA, 2003c)	
Aerator Properties (Not Used)				
d_imp	Impeller diameter (cm)	0	NA	SDU modeled as quiescent SI
F_aer	Fraction surface area-turbulent (fraction)	0	NA	
J	Oxygen transfer factor (lb O2/h-hp)	0	NA	
n_imp	Number of Impellers/aerators (dimensionless)	0	NA	
O2eff	Oxygen transfer correction factor (dimensionless)	0	NA	
Powr	Total Power for Impellers/aerators (hp)	0	NA	
w_imp	Impeller speed (rad/s)	0	NA	

B.5 Groundwater Model Inputs (EPACMTP)

Table B-7. EPACMTP Inputs

Parameter	Description & Units	Dry	Moderate	Wet	Reference	Comment
Vadose Zone Properties						
ALPHA	Moisture retention parameter (Van Genuchten) for unsaturated zone (1/cm)	0.07	0.009	0.016	FGD (US EPA, 2023b)	Median based on soil texture (ISTYPE1)
BETA	Moisture retention parameter for unsaturated zone (unitless)	1.885	1.236*	1.409	FGD (US EPA, 2023b), *except silty clay loam had no distribution in FGD, so median from Carsel and Parrish (1988)	Median based on soil texture (ISTYPE1)
DISPR	Longitudinal dispersivity in unsaturated zone (m)	0.21437	0.2884	0.10382	EPACMTP (US EPA, 2003c)	calculated from DSOIL using Eqn.5.2 in source
DSOIL	Depth from ground surface to water table (m)	8.835	12.2	3.81	Newell et al. (1990)	median
ISTYPE1	Soil type of vadose zone and aquifer	2 (Sandy Loam)	3 (Silty Clay Loam)	1 (Silty Loam)	SSURGO (USDA, 2016)	
POM	Percent organic matter in unsaturated zone (percent)	0.701	0.978	0.876	SSURGO (USDA, 2016)	mean within 3-mile radius; depends on soil texture (ISTYPE1)
RHOB	Bulk density of unsaturated zone soil (g/cm^3)	1.6	1.67	1.65	Carsel and Parrish (1988)	Depends on soil texture (ISTYPE1)
SATK	Saturated hydraulic conductivity of the unsaturated zone (cm/hr)	2.302	0.017	0.112	FGD (US EPA, 2023b)	Median for ash in fills; depends on soil texture (ISTYPE1)
WCR	Residual water content of the unsaturated zone (unitless)	0.065	0.089*	0.068	FGD (US EPA, 2023b), *except silty clay loam had no distribution in FGD, so median from Carsel and Parrish (1988)	Median based on soil texture (ISTYPE1)
WCS	Saturated water content (effective porosity) of the unsaturated zone (unitless)	0.41	0.43	0.45	Carsel and Parrish (1988)	Depends on soil texture (ISTYPE1)
Aquifer Properties						
Aquifer Type	Aquifer type	2 (Bedded Sed. Rock)	12 (Solution Limestone)	10 (Un- & Semi-consolidated Shallow Surficial Aquifers)	Newell et al. (1990)	
AL	Longitudinal dispersivity in the aquifer (m)	0.4437			EPACMTP (US EPA, 2003c)	estimated using Eqn. 5.11 in source and distance to well (XWELL) of 30 m, alpha_ref of 1 m
AT	Horizontal transverse dispersivity in the aquifer (m)	0.05546			EPACMTP (US EPA, 2003c)	estimated using Eqn. 5.13 in source (AL/8)

Parameter	Description & Units	Dry	Moderate	Wet	Reference	Comment
AV	Vertical transverse dispersivity in the aquifer (m)	0.002773			EPACMTP (US EPA, 2003c)	estimated using Eqn. 5.14 in source (AL/160)
BULKD	Aquifer soil bulk density (g/cm^3)	2.184	2.554	1.558	EPACMTP (US EPA, 2003c)	calculated from porosity using Eqn.5.6 in source; note Eqn. 5.6 has a typo; constant (which represents soil particle density) should 2.65 instead of 2.85†
FOC	Fraction of organic carbon in saturated soils (wt fraction)	0.004029	0.005621	0.005035	SSURGO (USDA, 2016)	calculated from POM of vadose zone (POM/174)
GRADNT	Regional hydraulic gradient in the aquifer (m/m)	0.0135	0.006	0.005	Newell et al. (1990)	median
POR	Volume fraction of connected pore space in the aquifer (unitless)	0.176	0.0363	0.412	Wolff (1982)	mean for aquifer type
TEMP	Ambient groundwater temperature (C)	9.6	12	19.4	Collins (1925)	
XKX	Hydraulic conductivity of saturated zone (aquifer) (m/yr)	252.5	1580	2295*	Newell et al. (1990)	median for aquifer type, *except Charleston [shallow surficial aquifer], where a mean value was used to avoid water table mounding
ZB	Thickness of saturated zone (m)	21.3	18.9	7.62	Newell et al. (1990)	median

† Eqn 5.6, as corrected for a particle density of 2.65 and using the variable names here, is BULKD = 2.65 (1-POR). Note that the porosities of some of the locations are very low due to the aquifer material, and hence the bulk density is relatively high compared to the particle density.

B.6 Surface Water Model Inputs (VVWM)

Table B-8. Standard Index Reservoir Parameters

Parameter	Description & Units	Value	Reference	Comment
Area_reserv	Reservoir area of the reservoir (m^2)	52,555	US EPA (2019a)	VVWM default
BNMAS	Reservoir benthic region areal concentration of biota (g/m^2)	0.006	US EPA (2019a)	VVWM default
bsp	Reservoir bed sediment porosity(fraction)	0.5	US EPA (2019a)	VVWM default
Bulk_density	Reservoir benthic region bulk density (g/mL)	1.85	US EPA (2019a)	VVWM default
CHL	Chlorophyll concentration in water column (mg/L)	0.005	US EPA (2019a)	VVWM default
D_over_dx_reserv	Mass transfer coefficient D/Δx (index reservoir) (m/s)	6E-09	US EPA (2019a)	VVWM default
db	Depth of upper benthic layer in reservoir (m)	0.05	US EPA (2019a)	VVWM default
DFAC	Photolysis parameter for reservoir	1.19	US EPA (2019a)	VVWM default
DOC1	Concentration of dissolved organic carbon in water column (mg/L)	5	US EPA (2019a)	VVWM default
DOC2	Concentration of dissolved organic carbon in benthic region (mg/L)	5	US EPA (2019a)	VVWM default
dwc_reservoir	Water column depth in the reservoir (m)	2.74	US EPA (2019a)	VVWM default
foc_bs (FROC2)	Fraction organic carbon for bed sediments (fraction)	0.04	US EPA (2019a)	VVWM default
foc_sw (FROC1)	Fraction organic carbon for suspended sediments (fraction)	0.04	US EPA (2019a)	VVWM default
PLMAS	Concentration of suspended biota (biomass) in water column (mg/L)	0.4	US EPA (2019a)	VVWM default
SUSED	Suspended solids concentration in water column (mg/L)	30	US EPA (2019a)	VVWM default

Table B-9. Other VVWM Inputs

Parameter	Description & Units	Value	Reference	Comment
Baseflow	Reservoir baseflow (m^3/s)	0	Assumption	
burialflag	Sediment burial flag: true = burial occurring and removing chemical	TRUE	NA	
Depth_0	Depth at which the input concentrations of physics parameters were measured for reservoir (m)	2.74	Set to the depth of the waterbody (dwc_reservoir)	
Depth_max	Maximum depth in the reservoir before overflow (m)	2.74	Set to the depth of the waterbody (dwc_reservoir)	
Flow_averaging	Number of days that are used to average the influent water in VVWM (#)	30	NA	
is_add_return_frequency	Is alternative return frequency to be used in addition to the 10-year return default for output?	FALSE	NA	
is_calc_prben	Is fraction of mass going to sediment calculated (TRUE) or prescribed by PRBEN (FALSE)?	TRUE	NA	

Table B-10. Unused Chemical-specific VVWM Parameters

Parameter	Description & Units	PFOA	PFOS	Reference	Comment
Heat_of_Henry	Enthalpy of phase transformation, aqueous to air solution [J/mol]	NA	NA	NA	This assessment does not include modeling transport through air
Kaer	Surface water column aerobic biodegradation rate (1/day)	0	0	NA	Based on PFOA/PFOS degradation literature
Kanaer	Sediment anaerobic degradation rate (1/day)	0	0	NA	
kh	Surface water hydrolysis rate (1/day)	0	0	NA	
Kpo	Surface water photolysis rate (1/day)	0	0	NA	
temp_ref_aer_all	Reference temperature for water column degradation (C)	NA	NA	NA	Not used as all degradation rates are zero
temp_ref_anae_all	Reference temperature for benthic degradation (C)	NA	NA	NA	

B.7 Food Chain Calculations

Table B-11. Plant Uptake Parameters

Parameter	Description & Units	Value	Reference	Comment
MAF_expfruit	Moisture adjustment factor for exposed fruit (% water)	85	EFH:2011 (US EPA, 2011)	Tables 9-37 (MAFs) and 13B-1 (crops assigned to categories). Average of MAFs for all crops in category
MAF_exveg	Moisture adjustment factor for exposed vegetables (% water)	90	EFH:2011 (US EPA, 2011)	Tables 9-37 (MAFs) and 13B-1 (crops assigned to categories). Average of MAFs for all crops in category
MAF_forage	Moisture adjustment factor for forage (% water)	80	MSU Extension (2011)	
MAF_grain	Moisture adjustment factor for grain (% water)	NA	NA	Not used: grain assumed to be uncontaminated; see Section 2.9.3.4
MAF_profruit	Moisture adjustment factor for protected fruit (% water)	87	EFH:2011 (US EPA, 2011)	Tables 9-37 (MAFs) and 13B-1 (crops assigned to categories). Average of MAFs for all crops in category
MAF_proveg	Moisture adjustment factor for protected vegetables (% water)	81	EFH:2011 (US EPA, 2011)	Tables 9-37 (MAFs) and 13B-1 (crops assigned to categories). Average of MAFs for all crops in category
MAF_root	Moisture adjustment factor for root vegetables (% water)	81	EFH:2011 (US EPA, 2011)	Tables 9-37 (MAFs) and 13B-1 (crops assigned to categories). Average of MAFs for all crops in category
MAF_silage	Moisture adjustment factor for silage (% water)	65	NDSU Extension (2021)	
VG_root	Empirical correction factor (root vegetables) (fraction)	1	HHRAP (US EPA, 2005)	Adjustment factor for high log Kow chemicals; Kow is not applicable to PFOA/PFOS
Unused Plant-Air Pathway Parameters				
Fw	Fraction of wet deposition adhering to plant surface (fraction)	NA	NA	The conceptual model for this assessment assumes no deposition or diffusion to plants
KpPar	Plant surface loss coefficient (particulate) (1/yr)	NA	NA	
Rp_[X]	Interception fraction (by category of aboveground plant) (fraction)	NA	NA	
td	Time period of deposition (yrs)	NA	NA	
Tp_[X]	Length of plant exposure to deposition (by category of above ground plant) (yrs)	NA	NA	
VG_[X]	Crop yield (by category of aboveground plant) (kg DW/m2)	NA	NA	
Yp_[X]	Empirical correction factor (by category of aboveground plant) (fraction)	NA	NA	

Table B-12. Livestock Exposure Parameters

Parameter	Description & Units	Value	Reference	Comment
Dairy Cows				
F_{forage}	Fraction of forage contaminated (fraction)	1	Assumption	
F_{grain}	Fraction of grain contaminated (fraction)	0	Assumption	Assumes all grain is uncontaminated
F_{silage}	Fraction of silage contaminated (fraction)	1	Assumption	
F_{water}	Fraction of water contaminated (fraction)	1	Assumption	
Q_{forage}	Quantity of forage consumed by livestock (kg DW/day)	13.2	HHRAP (US EPA, 2005)	
Q_{grain}	Quantity of grain consumed by livestock (kg DW/day)	3	HHRAP (US EPA, 2005)	
Q_{silage}	Quantity of silage consumed by livestock (kg DW/day)	4.1	HHRAP (US EPA, 2005)	
Q_{soil}	Quantity of soil consumed by livestock (kg/day)	0.4	HHRAP (US EPA, 2005)	
Q_{water}	Quantity of water consumed by livestock (L/day)	92	3MRA (US EPA, 2003b)	3MRA default
Beef Cattle				
F_{forage}	Fraction of forage contaminated (fraction)	1	Assumption	
F_{grain}	Fraction of grain contaminated (fraction)	0	Assumption	Assumes all grain is uncontaminated
F_{silage}	Fraction of silage contaminated (fraction)	1	Assumption	
F_{water}	Fraction of water contaminated (fraction)	1	Assumption	
Q_{forage}	Quantity of forage consumed by livestock (kg DW/day)	8.8	HHRAP (US EPA, 2005)	
Q_{grain}	Quantity of grain consumed by livestock (kg DW/day)	0.47	HHRAP (US EPA, 2005)	
Q_{silage}	Quantity of silage consumed by livestock (kg DW/day)	2.5	HHRAP (US EPA, 2005)	
Q_{soil}	Quantity of soil consumed by livestock (kg/day)	0.5	HHRAP (US EPA, 2005)	
Q_{water}	Quantity of water consumed by livestock (L/day)	53	3MRA (US EPA, 2003b)	3MRA default
Chickens (Laying Hens)				
F_{forage}	Fraction of forage contaminated (fraction)	1	Assumption	
F_{grain}	Fraction of grain contaminated (fraction)	0	Assumption	Assumes all grain is uncontaminated
F_{silage}	Fraction of silage contaminated (fraction)	1	Assumption	
F_{water}	Fraction of water contaminated (fraction)	1	Assumption	
Q_{forage}	Quantity of forage consumed by livestock (kg DW/day)	0.03	Dal Bosco et al. (2014)	
Q_{grain}	Quantity of silage consumed by livestock (kg DW/day)	0.074	Kowalczyk et al. (2020)	
Q_{silage}	Quantity of grain consumed by livestock (kg DW/day)	0.016	Kowalczyk et al. (2020)	
Q_{soil}	Quantity of soil consumed by livestock (kg/day)	0.02	HHRAP (US EPA, 2005)	
Q_{water}	Quantity of water consumed by livestock (L/day)	0.21	AECOM (2017)	

Table B-13. Transfer Factors for Food Chain Pathways

Parameter	Description & Units	PFOA	PFOS	Reference	Comment
Fish					
BAF_T3F	Bioaccumulation factor for trophic level 3 fish filet ([mg/kg fish]/[mg/L water])	49	1700	US EPA (2024a)	
BAF_T4F	Bioaccumulation factor for trophic level 4 fish filet ([mg/kg fish]/[mg/L water])	31	860	US EPA (2024a)	
Plants					
Br_Exfruit	Soil to plant uptake factor for exposed fruit ([mg/kg DW plant]/[mg/kg soil])	0.13	0.03	PFOA: Blaine et al. (2013, 2014); Lechner and Knapp (2011); PFOS: Blaine et al. (2014)	PFOA: median of tomatoes, sugar snap peas, cucumbers; pot studies; PFOS: sugar snap peas, pot study
Br_ExVeg	Soil to plant uptake factor for exposed vegetables ([mg/kg DW plant]/[mg/kg soil])	1.5	0.11	PFOA: Blaine et al. (2013, 2014); PFOS: Blaine et al. (2013)	PFOA: median of lettuce, celery; pot studies; PFOS: lettuce; field study
Br_Forage	Soil to plant uptake factor for forage ([mg/kg DW plant]/[mg/kg soil])	0.29	0.08	Yoo et al. (2011)	grass; field study
Br_Grain	Biotransfer factor (soil to grain) (mg/kg DW plant]/[mg/kg soil)	NA	NA	NA	Not used; all grain is assumed to be uncontaminated, see Section 2.9.3.4
Br_Profruit	Soil to plant uptake factor for protected fruit ([mg/kg DW plant]/[mg/kg soil])	0.13	0.03	PFOA: Blaine et al. (2013, 2014); Lechner and Knapp (2011); PFOS: Blaine et al. (2014)	PFOA: median of tomatoes, sugar snap peas, cucumbers; pot studies; PFOS: sugar snap peas, pot study
Br_Proveg	Soil to plant uptake factor for protected vegetables ([mg/kg DW plant]/[mg/kg soil])	1.5	0.11	PFOA: Blaine et al. (2013, 2014); PFOS: Blaine et al. (2013)	PFOA: median of lettuce, celery; pot studies; PFOS: lettuce; field study
Br_Root	Soil to plant uptake factor for root vegetables ([mg/kg DW plant]/[mg/kg soil])	0.73	0.8	Blaine et al. (2014); Lechner and Knapp (2011); Wen et al. (2016)	median of carrots, potatoes, radish; pot studies
Br_Silage	Soil to plant uptake factor for silage ([mg/kg DW plant]/[mg/kg soil])	0.29	0.08	Yoo et al. (2011)	grass; field study
Bv	Biotransfer factor (vapor phase air to plant) (ug/g DW plant]/[ug/g air)	NA	NA	NA	Volatilization is not expected under environmental conditions (see Section 2.2.2)
Animal Products					
Bs	Bioavailability of chemical in soil relative to plants (fraction)	1	1	HHRAP (US EPA, 2005)	Reflects the efficiency of transfer of contaminants from soil to livestock relative to transfer from plants to livestock. HHRAP cites inadequate data to set this to anything other than 1
BTF_beef	Biotransfer factor for beef ([mg/kg WW]/[kg DW/day])	0.01	0.18	PFOA: Vestergren et al. (2013) PFOS: Drew et al. (2021)	PFOA: Dairy cattle PFOS: Beef cattle
BTF_eggs	Biotransfer factor for eggs ([mg/kg WW]/[kg DW/day])	8.6	21	Wilson et al. (2021)	Laying hens
BTF_milk	Biotransfer factor for milk ([mg/kg WW]/[kg DW/day])	0.01	0.02	Vestergren et al. (2013)	Dairy cattle

Parameter	Description & Units	PFOA	PFOS	Reference	Comment
BTF_poultry	Biotransfer factor for chicken ([mg/kg WW]/[kg DW/day])	0.2	2.2	Kowalczyk et al. (2020)	Laying hens

B.8 Human Exposure Calculations

Table B-14. Exposure Factors

Parameter	Description & Units	Child (1-11yrs)	Adult Farmer	Reference	Comment
BW	Body weight (kg)	21	80	EFH:2011 (US EPA, 2011)	Table 8-1, mean (general population)
CR_beef	Daily human consumption rate of beef (g WW/kg BW/day)	2.1	1.6	EFH:2011 (US EPA, 2011)	Tbl 13-33, 50th percentile
CR_dw	Daily human consumption rate of water (mL/kg-day)	14	13.4	EFH:2019 drinking water update (US EPA, 2019b)	Tbl 3-21, 50th percentile
CR_eggs	Daily human consumption rate of eggs (g WW/kg BW/day)	0.7	0.7	EFH:2011 (US EPA, 2011)	Tbl 13-40, 50th percentile, households that farm (all ages)
CR_exfruit	Daily human consumption rate of exposed fruit (g WW/kg BW/day)	1.33	1.3	EFH:2011 (US EPA, 2011)	Tbl 13-58, 50th percentile
CR_exveg	Daily human consumption rate of exposed vegetables (g WW/kg BW/day)	1	1.4	EFH:2011 (US EPA, 2011)	Tbl 13-60, 50th percentile
CR_fish	Daily human consumption rate of fish (g WW/kg BW/day)	0.55	0.47	EFH:2011 (US EPA, 2011)	Tbl 13-20, 50th percentile, no data for 1-5, so based on 6-11; adult based on households that fish (all ages)
CR_milk	Daily human consumption rate of milk (g WW/kg BW/day)	22	12	Children: EFH: 2018 meat & dairy update (US EPA, 2018); Adult: EFH:2011 (US EPA, 2011)	Child: Tbl 11-4, 50th percentile; Adult: Tbl 13-25, 50th percentile
CR_poultry	Daily human consumption rate of poultry (g WW/kg BW/day)	2	1.1	Children: EFH: 2018 meat & dairy update (US EPA, 2018); Adult: EFH:2011 (US EPA, 2011)	Child: Tbl 11-6, mean; Adult: Tbl 13-52, 50th percentile, households that farm (all ages)
CR_profruit	Daily human consumption rate of protected fruit (g WW/kg BW/day)	2.3	2.1	EFH:2011 (US EPA, 2011)	Tbl 13-59, 50th percentile
CR_proveg	Daily human consumption rate of protected vegetables (g WW/kg BW/day)	1.1	0.6	EFH:2011 (US EPA, 2011)	Tbl 13-61, 50th percentile
CR_root	Daily human consumption rate of below ground vegetables (g WW/kg BW/day)	0.59	0.88	EFH:2011 (US EPA, 2011)	Tbl 13-62, 50th percentile
CRs	Daily human incidental soil ingestion rate (mg/day)	40	10	EFH:2017 soil update (US EPA, 2017b)	Table 5-1; data for soil only, which includes outdoor settled dust
F_i	Fraction of human diet item *i* contaminated (fraction)	1	1	Assumption	Assumes all food items in the category contaminated
F_T3	Fraction of fish consumed that is trophic level 3 (fraction)	0.14	0.14	EFH:2011 (US EPA, 2011)	Table 10-74

Parameter	Description & Units	Child (1-11yrs)	Adult Farmer	Reference	Comment
F_T4	Fraction of fish consumed that is trophic level 4 (fraction)	0.86	0.86	EFH:2011 (US EPA, 2011)	Table 10-74
L_i	Food preparation or cooking loss for diet item *i* (fraction)	0	0	Assumption	Assumes no cooking or prep losses

B.9 Risk Calculations

Table B-15. Cancer Dose Inputs

Parameter	Description & Units	Value	Reference	Comment
AT	Averaging time for cancer risk (yr)	70	RAGS Pt A (US EPA, 1989a)	
ED	Exposure duration (yr)	10	EFH:2011 (US EPA, 2011)	Based on residential mobility data, Tbl 16-113 (farmers), 50th percentile; also used for nearby residents: the 50th percentile for general population, all ages, from Table 16-108 is 9 yrs, so this is a reasonable value for nearby residents as well. Value used for children as well, assuming whole family has same exposure duration.
EF	Exposure frequency (day/yr)	350	Policy	

Table B-16. Toxicity Inputs

Parameter	Description & Units	PFOA	PFOS	Reference	Comment
CSForal	Oral cancer slope factor ([mg/kg/day]$^{-1}$)	29,300	39.5	US EPA (2024b)	Final PFOA-PFOS tox values
RfD	Reference dose (mg/kg/day)	3E-08	1E-07	US EPA (2024b)	Final PFOA-PFOS tox values

B.10 References

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APPENDIX C. GROUNDWATER MODELING

This section describes the refined groundwater modeling used to predict the fate and transport of PFOA/PFOS present in land-applied biosolids and biosolids disposed in surface disposal units to underlying soils and groundwater to determine impacts on drinking water wells that are connected to groundwater. **Sections C.1** and **C.2** provides a discussion on model selection, an overview of the conceptual model, including the basic approach and assumptions. **Section C.3** discusses the input parameters and values used in this risk analysis. **Section C.4** discusses the model outputs. **Appendix B** provides additional information about the inputs used in modeling the groundwater pathway using EPACMTP (US EPA, 2003a,b,d; 1997).

C.1 Model Selection

The groundwater pathway was modeled for this analysis to estimate the receptor concentrations that result from a predicted release of PFOA/PFOS from land-applied biosolids and sewage sludge disposed in surface disposal units. The release of PFOA/PFOS occurs when these pollutants in land-applied wastes or in sewage sludge stored in surface disposal units percolate through soils and into the subsurface. The releases of pollutant mass and infiltrating water were determined using waste management unit-specific models (land application unit, or LAU, and surface disposal unit, or SDU) developed for 3MRA, as described in assessment **Section 2.9**. These models generate time-series of pollutant mass fluxes and infiltrating water fluxes to the subsurface as well as releases to other exposure pathways, the latter a capability not available in the source terms provided in the groundwater model, EPACMTP. Therefore, to satisfy the multi-pathway analysis plan for this risk assessment, the 3MRA waste management unit models (LAU and SDU) are used to provide mass and water fluxes to EPACMTP for fate and transport simulations of the subsurface environment.

PFOA/PFOS in the land-applied wastes or leaching from sludge stored in surface disposal units are transported via aqueous-phase migration through the unsaturated zone (*i.e.*, the soil layer beneath the area of waste application and subsurface above the groundwater table) to the underlying saturated zone (*i.e.*, groundwater), and then down-gradient to a hypothetical residential drinking water well located 5 meters from the edge of the farm field (*i.e.*, center of the buffer). For this analysis, the exposure concentration was evaluated as the peak concentration at the intake point of the drinking water well (hereafter referred to as the receptor well). **Figure C-1** shows the conceptual model of the groundwater fate and the transport of contaminant releases from either a LAU or a SDU to a down-gradient receptor well with associated dilution and attenuation. Details about the modeled receptor well are provided later in this section.

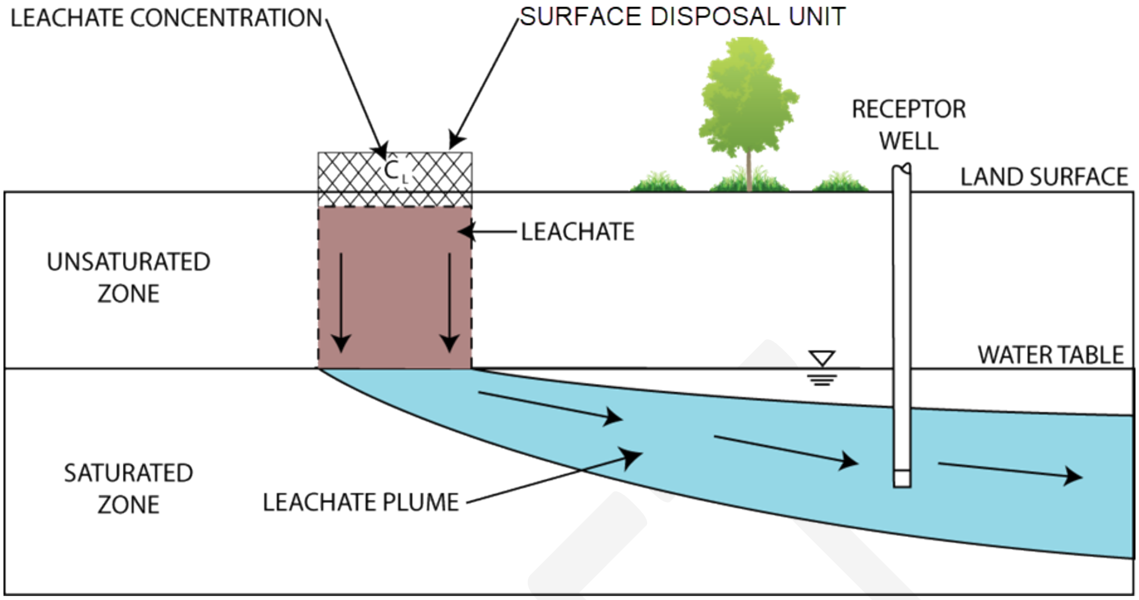


Figure C-1. Schematic diagram of groundwater modeling scenario.

C.1.1 Groundwater Model Selection

The mobility of PFAS in the environment, an active area of research, is known to be affected by their hydrophobic/hydrophilic-surfactant behavior (e.g., fluid-fluid interface retention), attraction to the solid phase in sediment (Higgins and Luthy, 2006; Liu et al., 2005), sludge (Milinovic et al., 2016), soil (Milinovic et al., 2015), to organic carbon in general (Higgins and Luthy, 2006), ionic behavior as a function of pH (Place and Field, 2012; Pereira et al. 2018), and the competition between these processes. Methodologies for assessing the impact of PFAS retention at the air-water interface (AWI) have been proposed (Brusseau, 2018; Zhang and Guo, 2024), modeled (Guelfo et al., 2020), and implemented in various fate and transport simulators (Guo et al., 2020; Silva et al., 2020; Guo et al. 2022).

Three simulation models were examined to determine which is best suited to support risk assessment objectives when assessing PFOA and PFOS:

- **EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP, US EPA, 2003a&b).** EPACMTP is EPA's conventional groundwater model and has been the traditional model used for both probabilistic and deterministic simulations of contaminant migration through the vadose zone to groundwater.
- **HYDRUS 1D with HD1 Pro Module (ver.5.01; Silva et al, 2020).** This is a deterministic model that includes a new AWI retention model developed specifically to address PFAS fate and transport. This model will be referred to as HYDRUS.
- **A recently published analytical PFAS leaching model (Guo et al., 2022).** This model includes some simplifying assumptions on the processes incorporated into the HYDRUS 1D PFAS module. This model will be referred to as ANALYTICAL.

Predictions of contaminant concentrations at the water table of an unconfined aquifer generally depend on two major processes within the vadose zone: flow and transport. For surfactants like PFOA and PFOS, transport processes that may occur when released into the subsurface include retention at the AWI, surfactant enhanced flow (e.g., Guo et al., 2020; Silva et al., 2020), self-assembly during sorption (e.g., Kalam et al., 2021), and enhanced transport of co-contaminants through emulsions (e.g., Kostarelos et al., 2021) and micelles (e.g., An et al., 2002; Simmons and McGuffin, 2007). Except for AWI, these other transport processes may be excluded on account of assuming relatively "small" PFAS concentrations in biosolids (e.g., formation of micelles) and exclusion of mixed wastes (e.g., transport of co-contaminants). **Table C-1** shows how each of the above models handles flow, transport and AWI processes.

Table C-1. Evaluated Models and How Major Processes Are Handled

Major Processes	EPACMTP	HYDRUS 1D w/ PFAS Module (HYDRUS)	Analytical PFAS Model (ANALYTICAL)
Flow	Steady state variable saturated flow	Steady state and transient variable saturated flow	Steady state unsaturated flow
Transport	Transient transport with linear equilibrium partitioning	Transient transport with linear equilibrium partitioning and AWI retention	Transient transport with equilibrium partitioning, AWI and kinetic solid-phase sorption
AWI	No	Yes	Yes

C.1.2 Approach to Model Selection Evaluation

The general approach to evaluating these models uses the land application unit (LAU) module (US EPA, 2003c) as a source term for unsaturated zone flow and transport simulations. The LAU module was developed to estimate annual average surface soil constituent concentrations and constituent mass release rates to the air, downslope land, and groundwater. The model simulates the vertical movement of pollutants within the agricultural land (releases through leaching to groundwater), volatile and particle releases to the air, and horizontal movement of pollutants (runoff and erosion from the agricultural land across any buffer area to a nearby waterbody). The model has the ability to consider losses from agricultural land due to hydrolysis and biodegradation, as well as leaching, volatilization, and particle emissions due to tilling (mixing) operations and wind erosion. LAU produces the following outputs resulting from land-applied biosolids to be used as inputs to the vadose zone models under consideration:

- Annual leach flux (g/m^2-day)
- Annual infiltration (m/day)
- Annual leachate concentrations (mg/L); these are computed as the ratio of the annual leach flux and the annual infiltration and used as input for the upper boundary condition for the transport models.

The LAU module can be used simulate both crop and pasture scenarios, which both reflect biosolids applied at an agronomic rate to a field and differ only in whether biosolids are tilled into the soil (crop) or not (pasture). This assessment uses the crop scenario for this evaluation as tilling reduces the amount of available contaminant mass to move off the field, maximizing the amount of mass that can leach to the subsurface.

C.1.3 Scenarios Selected for Model Evaluation

To fully evaluate differences among the models, EPA developed eight basic scenarios that reflect a broad range of key hydrogeologic conditions. Specifically, this assessment considered two different values that represent a range of national conditions for each of three parameters:

- Depth to water table
- Soil texture
- Meteorological setting.

***Depth to Water Table*.** This parameter, also called vadose zone thickness, defines the modeling region, so relatively short and long values were used to capture a range. Hydrogeologic environment data for national modeling of the groundwater pathway are sourced from the Hydrogeologic Database for Groundwater Modeling (Newell et al., 1990). **Figure C-2** summarizes those data for a variety of settings. Based on those data, we selected depths of 1 m and 10 m to capture roughly the second and third quartiles: this provides a range of values without including extreme values.

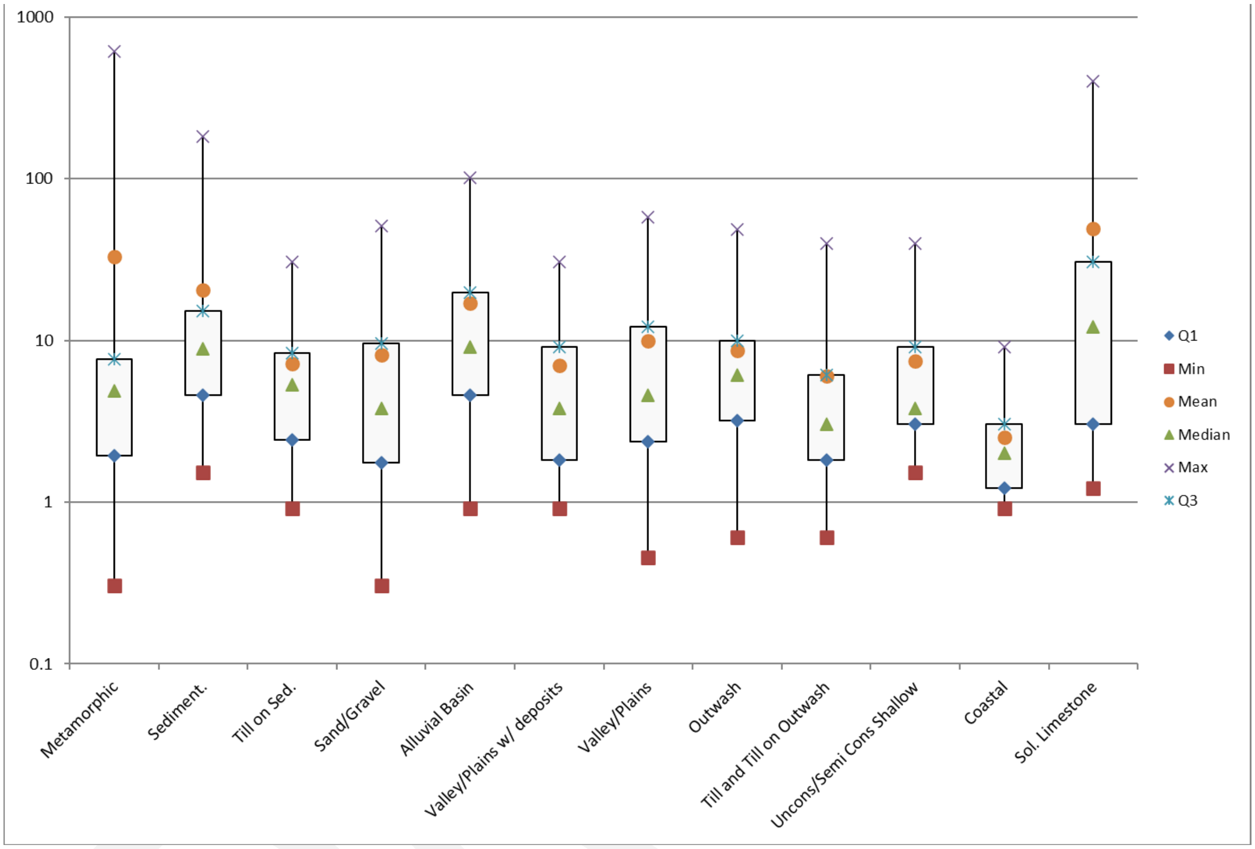


Figure C-2. Depth to water table data from HGDB.

Soil Texture. The relationship between infiltration (or pressure head) and the water content for a particular soil is known as the soil-water characteristic curve and is a key parameter needed to solve the governing flow equations in the unsaturated zone. These curves differ for different soil textures. Two reference soil texture types, loam and loamy sand, were chosen for model comparisons because they represent a broad range in saturated hydraulic conductivities that would likely result in significant contaminant mass transport to the water table; this helps evaluate the conservatism of each model and if the soil water characteristic curves used in the two models are similar. All three models evaluated use the empirical function proposed by Mualem (1976) and van Genuchten (1980) to estimate unsaturated hydraulic conductivity for both soil textures. This empirical function estimates the unsaturated hydraulic conductivity using the residual and saturated water contents (θr, θs) along with empirical Van Gneuchten parameters, α and β, that are obtained from characteristic soil-water retention curves for each soil type. **Table C-2** shows the values used for these soil properties; the same values were used for all three models evaluated.

Table C-2. Soil Parameter Values Used

Soil Column Property	Notation	Units	Loam	Loamy Sand
Depth to water table	-	m	1 or 10	1 or 10
Residual water content	θr	dimensionless	0.078	0.057
Saturated water content	θs	dimensionless	0.43	0.41
Saturated hydraulic conductivity	Ks	cm/hr	1.04	14.59
Van Genuchten parameter	α	cm-1	0.036	0.124
Van Genuchten parameter	β	dimensionless	1.56	2.28
Bulk Density	ρ	g/cc	1.33	1.65
Dispersivity	α_L	m	0.1 or 1	0.1 or 1
Percent organic matter	%OM	dimensionless	0.174	0.174
Fraction organic carbon	foc	dimensionless	0.001	0.001

Meteorological Setting. Both models simulate the soil water content as a function of infiltration (or pressure head) using the Van Genuchten model (1980) but using two infiltration scenarios can help evaluate whether both models simulate long term average flow conditions similarly given varying infiltration or recharge inputs. This assessment uses three meteorological settings: wet, moderate, and dry. For this evaluation, we used the wet and dry settings, as they represent bounding conditions. Ten years of meteorologic data from Charleston, SC, and Boulder, CO, are cycled 15 times to represent wet and dry meteorology, respectively, for 150-year simulations. In summary, the eight basic scenarios are presented **Table C-3**.

Table C-3. Modeling Scenarios

Depth to Water Table	Soil Type	Meteorological Conditions
1 m	Loam	Wet
		Dry
	Loamy Sand	Wet
		Dry
10 m	Loam	Wet
		Dry
	Loamy Sand	Wet
		Dry

EPACMTP can simulate only linear, instantaneous solid-phase adsorption; HYDRUS and ANALYTICAL can be run only assuming instantaneous and kinetic effects of adsorption (no AWI effects) or also including AWI effects. While HYDRUS can only model instantaneous linear solid-phase adsorption, ANALYTICAL can model both instantaneous and kinetic effects of adsorption. Kinetics associated with solid-phase adsorption were shown to be present in both batch and miscible -displacement experiments. Further, both HYDRUS and ANALYTICAL can model AWI effects using different values of the equilibrium distribution constant between the liquid phase and air-water interface (K_h). HYDRUS and ANALYTICAL were run assuming linear, solid-phase adsorption mode and with and without AWI effects for three different values of K_h, however, the input specifications required to simulate AWI effects differ between the HYDRUS and ANALYTICAL models: HYDRUS calculates K_h using a Langmuir approach whereas K_h is directly specified in ANALYTICAL.

Constituent Transport Parameters. The models require various constituent-specific transport factors. **Table C-4** presents the values used for PFOA and PFOS and indicates which of the three models evaluated use them.

Table C-4. Constituent Transport Parameters Including AWI and Sources

			Used in Models			Values		
Chemical Property	**Notation**	**Units**	**EPACMTP**	**HYDRUS**	**ANALYTICAL**	**PFOA**	**PFOS**	**Reference/Notes**
Diffusion coefficient in water	Diff in H2O	m^2/yr	●	●	●	0	0	
Organic partition coefficient	Koc	mL/g	●	●	●	1.99E+03	1.86E+04	Silva et al., 2020
Solid-phase (instantaneous) adsorption coefficient	Kd	mL/g	●	●	●	1.99	18.60	Silva et al., 2020
Langmuir adsorption isotherm maximum interfacial adsorbed (AWI) concentration	Tmax	mol/cm^2	—	●	—	5.54E-07	3.50E-07	Only applicable for HYDRUS (Silva et al., 2020). Kh_AWI directly specified in Analytical model.
Langmuir coefficient for AWI adsorption	KL_aw	cm^3/mol	—	●	—	6.67E+03	1.37E+05	Silva et al., 2020. Kh_AWI directly specified in Analytical model.
Fraction of sorbent for which sorption is instantaneous	Fs	fraction	—	—	●			Guo et al., 2022 (only applicable to two-domain solid-phase sorption models)
First order rate constant for kinetic sorption	αs		—	—	●			
Equilibrium distribution constant between liquid phase and air-water interface	Kh_AWI	cm	—	●	●	3.69E-03	4.79E-02	Silva et al., 2020 calculated Kh_AWI =KL_AWI*Tmax
Scaling constant to linearly scale the interfacial area	Scal_AWI	-	—	●	●	1	1	
Langmuir air-water interface sorption parameter	Nu_AWI	m^3/g	—	●	●	0	0	Set equal to zero if Langmuir sorption to the air-water interface is not to be considered
Non-linear (Freundlich) sorption Coefficient, β, for material type.	Beta_AWI	-	—	●	●	1	1	Set equal to one since Freundlich sorption to the air-water interface is not to considered.

● = used
— = not used

Boundary Conditions. The models require different types of upper and lower boundary conditions for flow and transport. **Table C-5** presents the types of flow and transport boundary conditions used by each of the models.

Table C-5. Boundary Conditions – Flow and Transport

Model	**Upper Boundary Condition**	**Lower Boundary Condition**
Flow		
HYDRUS	Variable Pressure Head/Flux	Free Drainage/Zero Pressure Gradient
EPACMTP	Constant Flux	Constant Pressure Head
ANALYTICAL	Constant Pressure Head/Flux	Free Drainage/Zero Pressure Gradient
Transport		
HYDRUS	Constant Mass Flux	Zero Concentration Gradient

Model	Upper Boundary Condition	Lower Boundary Condition
EPACMTP	Constant Mass Flux	Zero Concentration Gradient
ANALYTICAL	Constant Mass Flux	Zero Concentration Gradient

Model Simulation Parameters – Time Steps, Duration, Discretization. The process by which most physics-based simulators generate predictions of contaminant concentration in space and time is through the partitioning of both dimensions into small, discrete segments, and then repeatedly solving one or more equations of state or mass conservation across each spatial compartment in the model domain for each increment of time for some duration. The selection of an incremental space (*i.e.*, distance, area, volume) and time for a simulation is dependent in part on a simulator's numerical approach to solving the physics-based equations, the modeling objective, and balancing the need for accuracy and computational effort. In general, for porous media flow and transport, the spatial domain is divided into computational cells or nodes that are small enough to capture the spatial variability of the state variable (*e.g.*, saturation, pressure, dissolved concentration) in the region of interest at any point in time, and in small enough time increments to capture key changes in the state variable, like the peak elevation or concentration. Finally, the number of time increments to evaluate should be sufficient to capture the temporal variability of the process in the region of interest. For these model comparisons, the objective is to evaluate the arrival and dissipation of the contaminant at the water table. **Table C-6** presents the spatial and temporal discretization parameters used for the simulators and scenarios conducted in this comparison.

For HYDRUS and the ANALYTICAL model, space and time discretization is prescriptive – the modeler must select these parameters. In EPACMTP, time and space discretization are internally determined and optimized to accurately capture water table concentrations for thousands of Monte Carlo simulations. Spatial discretization is finer near the water table to capture the region of the unsaturated zone where saturation changes most rapidly. The number of time steps are fixed but sufficient to capture the arrival and dissipation of the contaminant front at the water table. As the model domains examined here are small, computational burden is not an issue. Therefore, rectilinear grid cells for both HYDRUS and the ANALYTICAL model were specified as 1 cubic centimeter and concentration predictions were generated daily at the water table. Simulation durations in all models were dictated by the combination of slow advection in the dry environment and high retardation based on K_d and the objective of capturing the entire concentration breakthrough at the water table.

Table C-6. Spatial and Temporal Simulation Parameters

Model	EPACMTP	HYDRUS	ANALYTICAL*
Spatial Discretization of Unsaturated Zone	Computational points are automatically established at 0.0, 0.6, 0.75, 0.85, 0.95, and 1.0 x depth to water table in meters	1 cm grid cells were specified to discretize the depth to water table for both 1 m and 10 m scenarios	1 cm grid cells were specified to discretize the depth to water table for both 1 m and 10 m scenarios[1]
Temporal Discretization	3000 equal timesteps are automatically determined between an estimated arrival and dissipation time of the concentration front at the water table in years	Daily timesteps were specified	Daily timesteps were specified
Simulation Duration	10,000 and 20,000 years were specified for wet and dry scenarios, respectively	10,000 and 20,000 years were specified for wet and dry scenarios, respectively	10,000 and 20,000 years were specified for wet and dry scenarios, respectively

* Spatial and temporal discretization of the ANALYTICAL model are not used for computing numerical solution but for data presentation purposes only.

LAU Outputs Used as Inputs to Vadose Zone Models

Crop scenario simulations conducted with the LAU module for biosolids containing PFOA and PFOS applied in wet and dry environs were used to create time series of mass and water fluxes to represent the leaching of these contaminants from land applied biosolids. **Figure C-3** shows resulting mass fluxes (left-hand vertical axes) and water fluxes (right-hand vertical axes) for PFOS (top) and PFOA (bottom) in wet (orange data points) and dry (yellow data points) environments. The cyclic nature of these fluxes reflects the cyclic meteorological datasets. **Figure C-4** shows the resulting leachate concentration over time for PFOS and PFOA in the two meteorologic settings. Leachate concentration was calculated by dividing the mass flux by the water flux for each time point. These plots show that constant annual concentrations in leachate are generated from surface soils during the 40-year period of biosolids application. In the case of PFOA, concentrations drop off after the 40-year period reflecting no additional mass and the dissolution of residual PFOA sorbed to soils. For PFOS, the leachate concentration does not change much over time. This is attributed to the high K_{oc} value limiting the amount of dissolvable mass to infiltrating water and that the reservoir of sorbed mass is enough to maintain the limited available mass for a longer time.

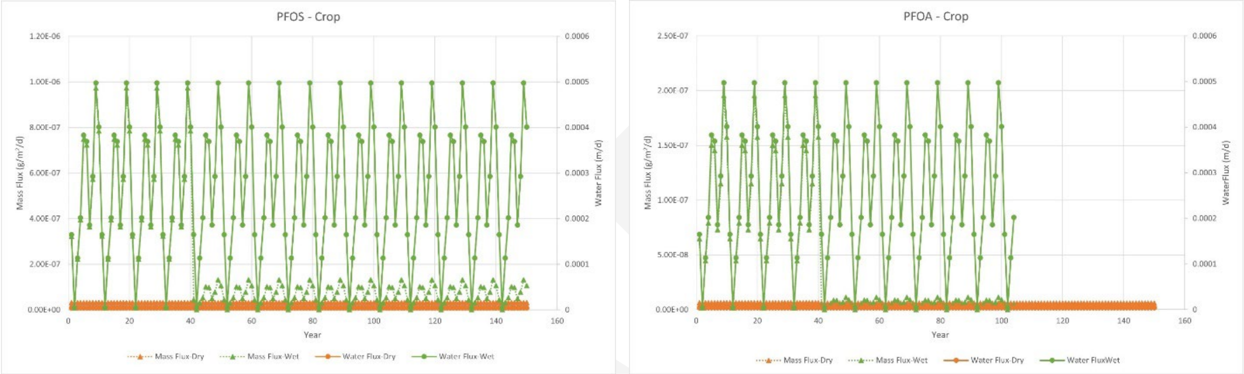


Figure C-3. Leachate flux for PFOS (left) and PFOA (right) for crop scenario.

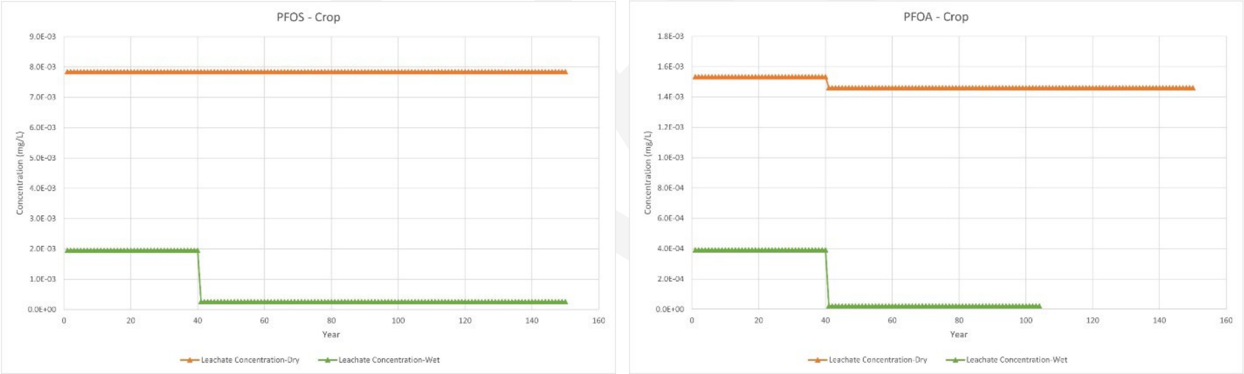


Figure C-4. Leachate concentration for PFOS (left) and PFOA (right) for crop scenario.

For the purposes of model comparison, all three models were subject to constant water flux (infiltration rate) at the top of the soil column equal to the average water flux over the 150-year simulation for wet and dry scenarios. Likewise, constant leachate concentrations for each constituent-meteorology combination during the 40-year application period were used to define the transport boundary condition at the top of each model. Modeled values for infiltration and leachate concentrations are presented in **Table C-7**.

Table C-7. Boundary Conditions – Infiltration Rates and Leachate Concentrations

Boundary Condition	PFOA	PFOS
Infiltration [m/yr]		
Wet Meteorology	9.33E-2	
Dry Meteorology	1.49E-3	
Leachate Concentration [mg/L]		
Wet Meteorology	3.93E-4	1.96E-3
Dry Meteorology	1.53E-3	7.85E-3

C.1.4 Model Selection Evaluation Results

This section presents modeling results from EPACMTP, ANALYTICAL and HYDRUS for the various unsaturated zone model scenarios described above. Results for groundwater flow within the vadose zone are presented first to evaluate whether both models can simulate similar water content profiles within the vadose zone for the same set of initial and boundary conditions. If both models simulate similar water content profiles, differences in PFOA/PFOS concentration results from the transport simulations, both with and without AWI effects can be inferred to be due to differences in how each model handles PFOA/PFOS transport.

C.1.4.1 Flow

Groundwater flow results from EPACMTP, ANALYTICAL and HYDRUS for the various model scenarios were compared using the simulation of volumetric water content profiles at steady state within the vadose zone. The volumetric water content describes the volume of water per unit volume of soil, generally expressed as a dimensionless fraction or percentage. Comparing these profiles would illuminate any differences between the mathematical formulations used in simulating unsaturated zone flow. As described earlier, the governing flow equation in EPACMTP is given by Darcy's law, a steady-state infiltration is used in ANALYTICAL model, while HYDRUS uses a modified form of the Richards equation. Note these profiles are not constituent specific.

Figure C-5 presents steady-state water content profiles from EPACMTP/HYDRUS for a 1-m (top) and 10-m (bottom) soil column under wet (left) and dry (right) conditions for loam and loamy sand, assuming the same boundary conditions. Note that the ANALYTICAL model is not used in this initial comparison since the model assumptions lend to a single computed value of water content instead of a depth-dependent profile.

For the 1-m soil column (Figure C-5, top), there is little difference between the HYDRUS and EPACMTP models for any given soil texture or infiltration condition, and for both models, the simulated water content profiles are very similar wet and dry conditions given the same soil texture: the maximum difference is approximately 2%. These similar water content profiles for varying infiltration scenarios suggests that the default parameters in the Van Genuchten (1980) model used by both models to simulate the soil water content as a function of infiltration (or pressure head) are compatible and can be used to simulate steady state conditions for a given infiltration input.

Conversely, the differences between predicted water content profiles between the two soil textures (for the same infiltration and model) are larger, 8–22% between loam and loamy sand under wet scenario for HYDRUS (Figure C-5). This can be attributed to differences in the soil water characteristic curves for the two soil textures. Similar water content profiles simulated by both models for each soil texture suggests that the parameters used in the soil water characteristic curves to solve the differing governing flow equations used in both models do not have a significant impact on the predicted water contents.

For the 10-m soil column (Figure C-5, bottom), we see similar results for infiltration scenarios: differences between dry and wet scenarios for a specified soil texture are less than 2% for all depths except very close to the water table, between 9 and 10 meters, where the difference between the predicted water contents from the two models is up to 15%. At a depth of 9 meters, HYDRUS first predicts lower water contents by up to 6% as compared to EPACMTP and then predicts higher water contents than EPACMTP closer to the water table at 10 meters, with the maximum difference of 15% at 9.5 meters for loamy sand, dry infiltration scenario. Since these differences between the two models are only observed close to the water table, it may suggest discrepancies in interpretation of the water table boundary condition by the two models. This may also be an indication that for a deeper vadose zone column, the differing mathematical formulations governing flow used in the two models (Darcy's law vs. Richard's equation), may show an observed influence on water content profiles, particularly close to the water table.

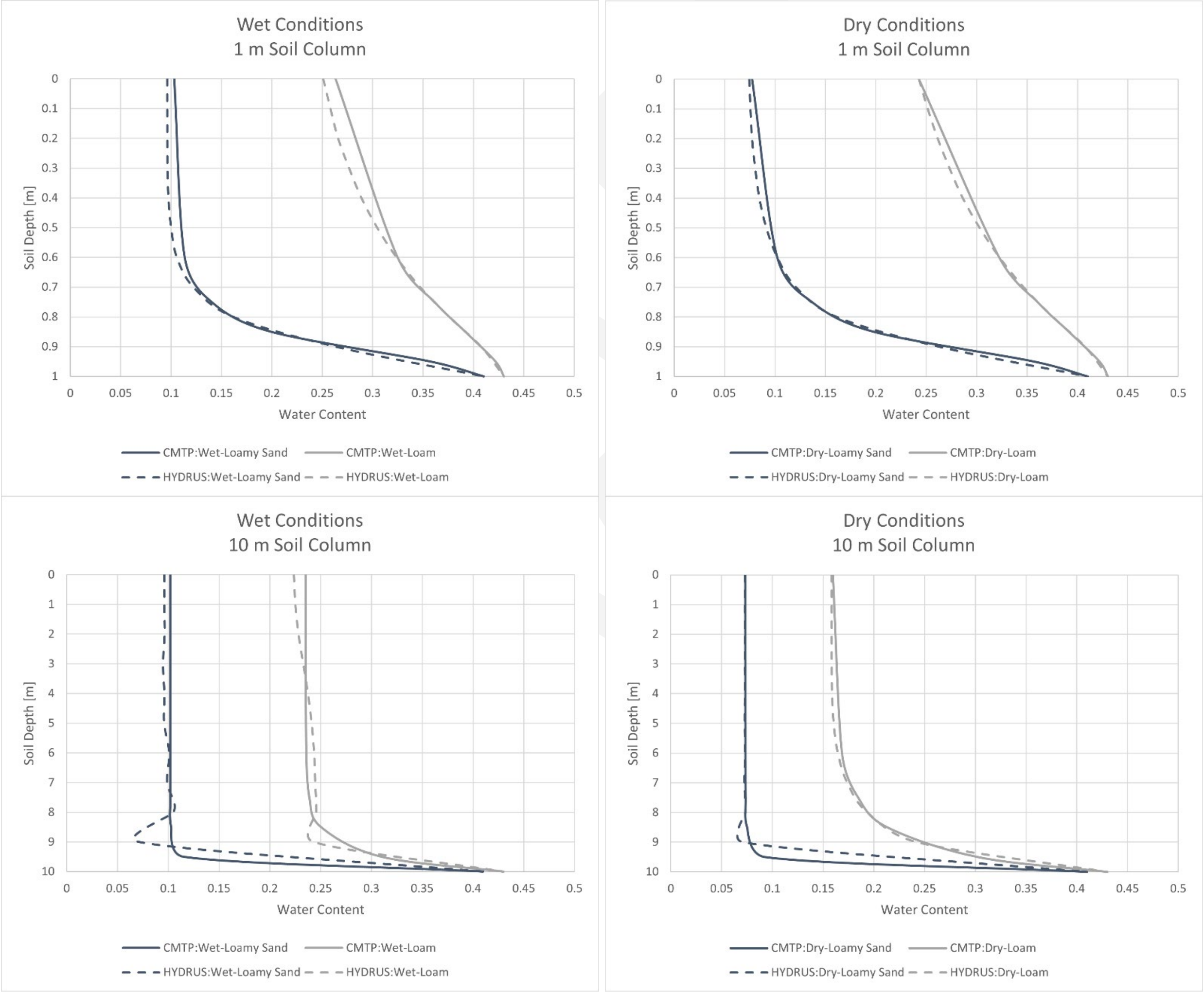


Figure C-5. **Water content profiles for 1-m (top) and 10-m (bottom) soil column for loam and loamy sand under wet (left) and dry (right) scenarios simulated using EPACMTP (solid lines) and HYDRUS (dashed lines).**

Even though EPACMTP computes a variably saturated soil profile that compares well with HYDRUS under the same boundary conditions (Figure 4), EPACMTP uses a depth-averaged water content for the analytical transport solution. This is a very useful technique when running several thousand model runs under a probabilistic framework. The ANALYTICAL model also uses a singular water content value for the analytical transport solution. However, a direct comparison of the water content values between the EPACMTP and ANALYTICAL models would not be correct since both models assume different boundary conditions at the water table (*i.e.*, constant pressure head vs. free drainage). However, the boundary conditions in HYDRUS can be changed to free drainage for comparison with the ANALYTICAL model even though HYDRUS uses the variable water content profile shown in Figure C-5 for its transport solutions. **Figure C-6** presents the steady-state water content profiles from the ANALYTICAL model (blue bar) and HYDRUS (orange bar) for a 1-m and 10-m soil column under wet and dry conditions for loam and loamy sand. From Figure C-6, it can be observed that the simulated water contents from the ANALYTICAL and HYDRUS models are very similar for every scenario tested under the same boundary conditions (blue and orange bars).

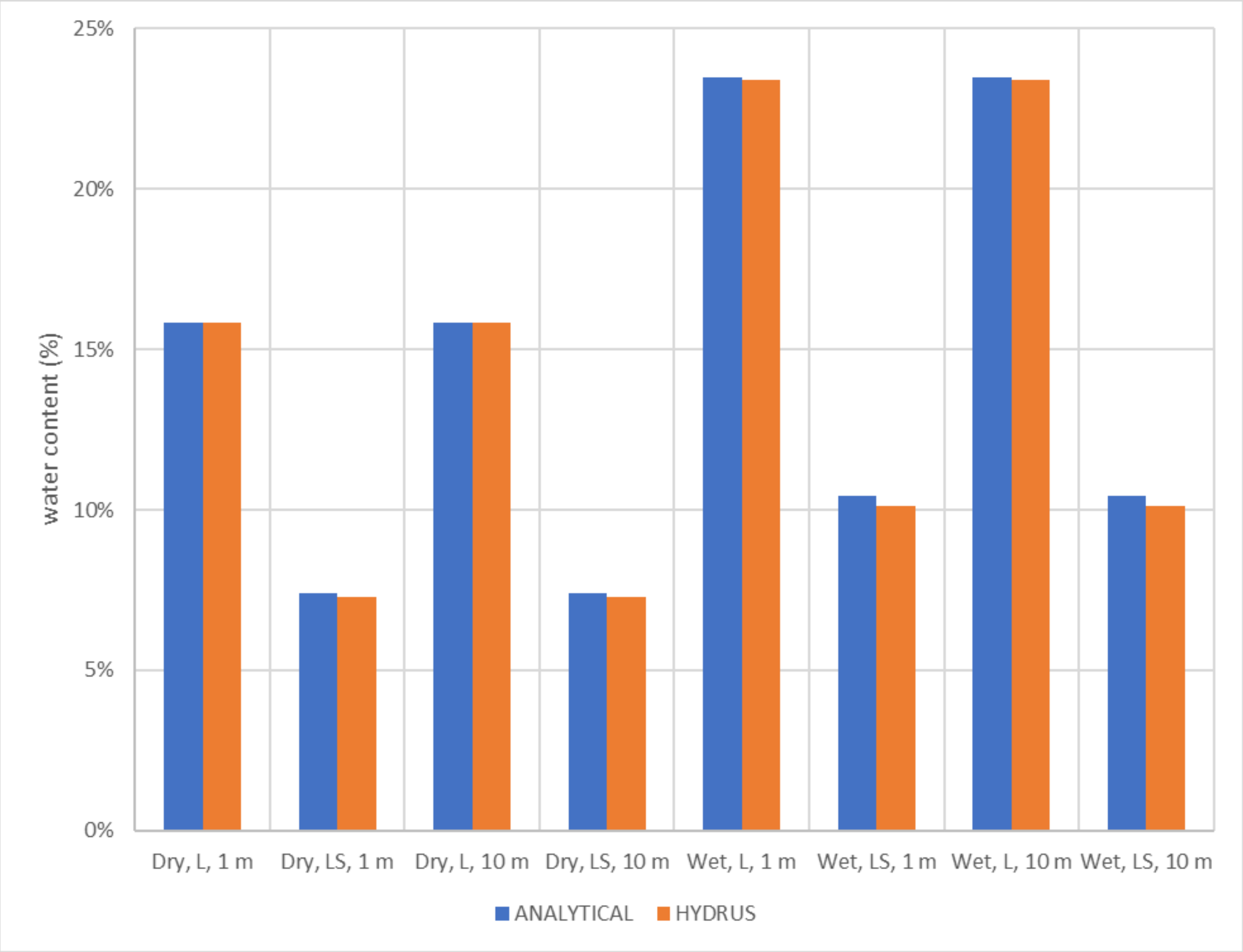


Figure C-6. Water content profiles for 1-m and 10-m soil column for loam and loamy sand under wet and dry scenarios simulated using HYDRUS and ANALYTICAL models.

Overall, the simulation results shown in Figures C-5 and C-6 confirm that there is little difference between three models in simulating variable saturated flow regardless of soil textures, meteorological environments, vadose zone thickness. However, for a deeper vadose zone soil column, the influence of water table boundary conditions and governing flow equations on simulated water content profiles should be carefully considered.

C.1.4.2 Transport

The figures in this section present breakthrough curves from EPACMTP, ANALYTICAL and HYDRUS models for PFOA (top) and PFOS (bottom) for the different soil column depths and meteorological conditions. For all figures, both models were run using linear adsorption (no AWI effects; solid lines) and both the ANALYTICAL model and HYDRUS were run with AWI effects (dashed lines) using three values of the equilibrium distribution constant between the liquid phase and air-water interface (K_{aw}). Note that both the ANALYTICAL model and the most recent version of HYDRUS (version 5.01) used in this testing can only simulate instantaneous, equilibrium sorption at the AWI. Recent column experiments have shown that kinetics associated with AWI adsorption is minimal under steady-state flow conditions (Brusseau, 2020; Brusseau et al., 2021).

Shallow Water Table, Wet Environment

For a 1-m soil column in a wet environment (**Figure C-7**), breakthrough curves simulated by the EPACMTP, ANALYTICAL and HYDRUS models show excellent agreement for both soil types and constituents in terms of their magnitude and peak arrival time when considering only linear, solid-phase sorption.

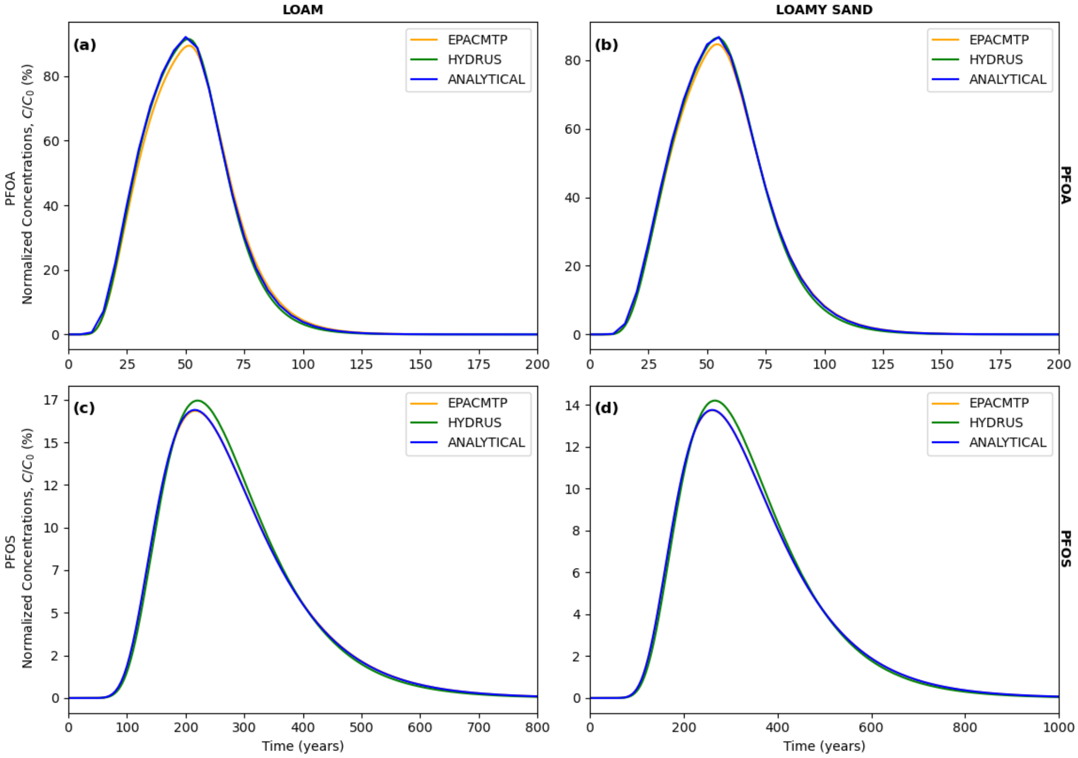


Figure C-7. Breakthrough curves at a 1-m water table depth in a wet environment using linear sorption.

Note that for the same soil type and meteorological condition, the peak magnitude observed at the water table for PFOA is much greater than that for PFOS (peak concentrations >80% of input concentrations for PFOA vs 14-17% for PFOS). This lower peak magnitude observed at the water table for PFOS is due to stronger solid phase adsorption of PFOS to soil organics and is reflected by the difference in their representative K_d value chosen for the modeling effort (see Table C-4). When comparing the simulated peak magnitudes for the same constituent but across soil types, the peaks for Loam are slightly greater than that for Loamy Sand (approximately less than 3%). These differences are likely due to the higher calculated dispersion coefficients for Loamy Sand as compared to Loam. The dispersion coefficients in all three models are calculated as:

$$D = v \times \alpha_L = \left(\frac{q}{\theta}\right) \times \alpha_L$$

where,

D = dispersion coefficient (L^2T^{-1} such as cm^2/s)

V = pore velocity or Darcy velocity (LT^{-1} such as cm/s)

Q = infiltration rate (LT^{-1} such as cm/s)

N = model-calculated water content (L^2/L^3 such as cm^2/cm^3)

α_L = dispersivity (L such as cm).

While the infiltration rate (q) and the dispersivity (α_L) are the same for both soil types, the model computed pore velocities (v) and water contents (ϑ) are different owing to the differences in their saturated hydraulic conductivities input to the model. The pore velocities computed for Loamy Sand are higher than that for Loam while the water content for Loam is higher than Loamy Sand. For example, in the wet scenario, 1 m soil column, the ANALYTICAL model calculated pore velocity for Loam at 1.26×10^{-6} cm/s corresponding to a water content of ~24.5%. For Loamy Sand, the calculated pore velocity was 2.84×10^{-6} cm/s corresponding to a water content of ~10.4%. Since the dispersion coefficient is directly proportional to the pore velocity (or inversely proportional to the water content), the higher pore velocity for Loamy Sand may have contributed to increased dispersion and produced the slightly lower peak observed.

The simulated PFOA/PFOS arrival time at the water table based on the breakthrough curves (Figure C-7) are only illustrative for model comparison purposes and may appear greater than those observed in field studies. For instance, modeled peak values of PFOS arrive at the water table in a Loamy Sand column under a wet scenario in approximately 216 years. This might appear contradictory to field observations of PFOS observed at the water in deeper soil columns. There may be several reasons to note regarding differences between model results and field observations:

- In the models, the modeled peak arrival times are a direct function of the representative K_d value chosen for the model comparison simulations (see Table C-4). That is, lower values of K_d chosen, the earlier the modeled peak arrival time. This can be illustrated by selecting an extremely low value of K_d=0 (*i.e.,* considering only advection and dispersion). In this scenario, PFOS is simulated to arrive at the water table in 4.4 years. K_d values for PFAS span a large range generally due to the varying soil types, field or laboratory conditions under which they were measured. A single K_d, organic partitioning coefficient (K_{oc}) and fraction organic carbon (f_{oc}), values were chosen for each of PFOA/PFOS (see Tables C-2 and C-4). However, the range of K_{oc} reported in literature is large and can often span several orders of magnitude. There is significant ongoing research on refining the K_{oc} values and understanding the partitioning behavior of PFAS in the environment, which can also depend on site-specific factors that are not considered here. When modeling site-specific observations, K_d should be carefully considered prior to making model comparisons to field observations. For the experiments reported here, the K_d values for each constituent was kept constant throughout the simulations for an appropriate comparison of results from different models.
- Models tested here do not account for preferential flow paths for PFAS migration to the water table due to soil heterogeneity that maybe present under field conditions. For example, Zeng and Guo (2021) have shown that preferential flow pathways generated by soil

heterogeneities can destroy air-water interfaces that can lead to early arrival and accelerated leaching of (especially long-chain) PFAS.

- Source leachate concentrations used in this modeling exercise are less than 0.01 mg/L (see Table C7), which is likely several orders of magnitude lower than source concentrations often reported in several studies (Anderson et al., 2016).

Including the effects of AWI (**Figure C-8**), we see that the ANALYTICAL and HYDRUS models show excellent agreement for both soil types and constituents in terms of their magnitude and peak arrival time. Generally, it is observed for both models that AWI decreases the magnitude of the peak concentrations and increases the arrival time of the peak at the water table for both PFOA and PFOS compared to scenarios considering only linear, solid-phase sorption. The higher the interfacial adsorption coefficient at the AWI, the more pronounced the effects on peak concentrations and arrival times. This is consistent with increased retardation of PFOA/PFOS anticipated with retention at the AWI. In addition to the difference in peak magnitude noted between PFOA/PFOS, the tail of the breakthrough curve is much longer for PFOS in comparison to PFOA. The longer breakthrough curve tail observed for PFOS is likely owing to its stronger retention to the solid phase. Another interesting observation is that when including the effects of AWI, the peak magnitude observed for Loamy Sand is higher than that for Loam within each constituent (PFOA or PFOS) and K_{aw} value. This is the opposite of what was observed when only considering solid phase sorption, when the peak magnitude for Loam was higher than Loamy Sand. The higher peak magnitude observed for Loamy Sand as compared to Loam is likely because there is less AWI adsorption for Loamy Sand than Loam. The reason for this is that the model computed total air water interfacial area is lower for Loamy Sand (~55 cm^2/cm^3) than Loam (~112 cm^2/cm^3), which leads to reduced AWI adsorption and lower retention of PFAS within the vadose zone of a Loamy Sand column than Loam.

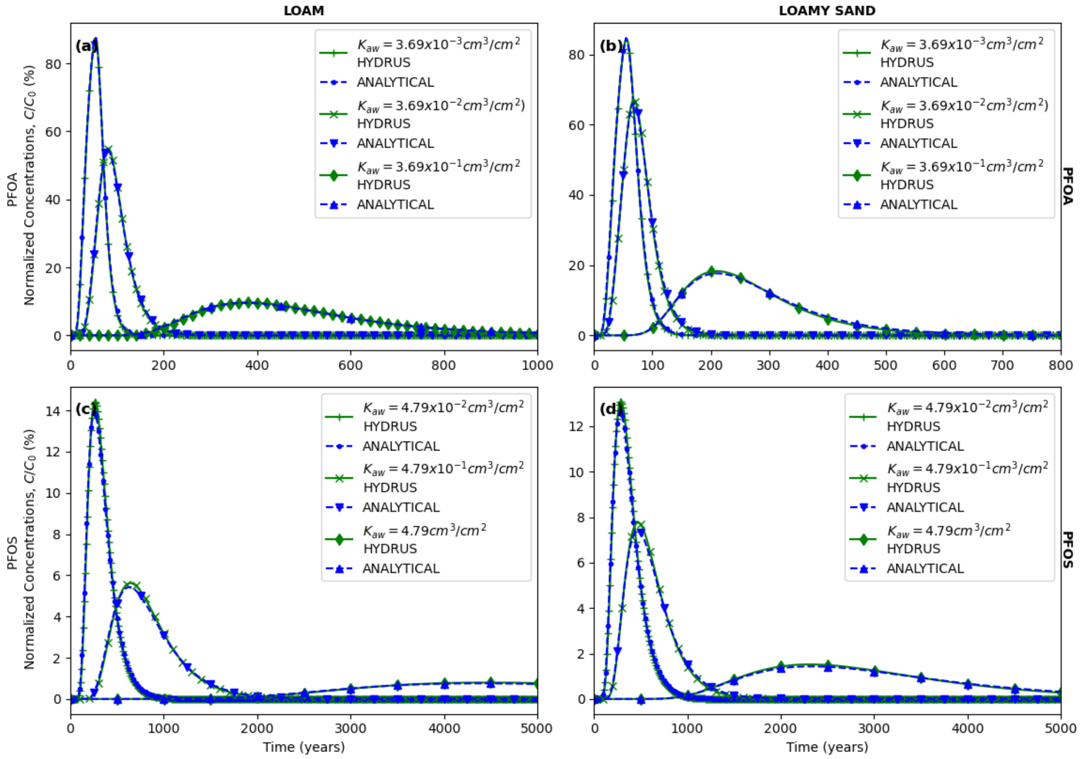


Figure C-8. Breakthrough curves at a 1-m water table depth in a wet environment using linear sorption and AWI effects for three values of Kaw (Silva et al., 2020).

However, some recent studies based on field data have shown a negative association between clay content and PFAS migration to the water table (see e.g., Andersen et al., 2019). Using data from US Air Force sites, these authors have shown that soils with higher clay content show statistically significant lower soil retention (or higher groundwater concentrations) than more permeable soils. Andersen et al. (2019) suggested three possibilities to explain their observation:

1. Relatively lower clay content soils are better drained and less prone to saturation during precipitation events. Lower water saturation would lead to higher magnitude of air-water interfacial area, and therefore retardation (Peng and Brusseau, 2005).
2. Electrostatic interactions between the negatively charged clay minerals and anionic PFAS may enhance transport to the water table due to anionic repulsion (Wang et al., 2015).
3. Soils with higher clay content retain relatively larger volumetric water content following precipitation events resulting in longer reaction time between aqueous and adsorbed PFAS and thus, kinetic-limited PFAS sorption (Wei et al., 2017), which would promote greater partitioning in higher clay content soils.

While the findings of Andersen et al. (2019) may seem contradictory to the modeled results presented here, one important thing to note is that all the model soil columns are forced with the same infiltration rate to make even comparisons for the purposes of this report. At field sites, the infiltration rate is likely to vary by soil type owing to the differences in their water retention capacities and surface evaporation rates.

Shallow Water Table, Dry Environment

For a 1-m soil column in a dry environment (**Figure C-9**), the breakthrough curves for EPACMTP, ANALYTICAL and HYDRUS under linear, solid phase sorption suggest PFOA/PFOS mass is strongly adsorbed to the soil and very little mass reaches the water table (<0.27% for PFOS and <2.5% for PFOA), even though the total mass of PFAS applied are the same for both dry and wet environments. These results suggest that in the dry environment there is much less advective/dispersive transport of PFOA/PFOS to the water table. In the model, this is evidenced by the calculated pore velocities and dispersion coefficients that are 2 orders of magnitude lower for the dry scenario as compared to the wet scenario. There is also slightly stronger solid-phase adsorption calculated in the dry scenario as compared to the wet scenario because of a higher retardation factor under the dry scenario. This is due to solid phase retardation is inversely proportional to the soil water content, which is approximately 12.7% for a wet, Loamy Sand, 1m column and 4.6% for a dry Loamy Sand, 1 m column. These observations are consistent with studies that used field data from many sites and showed that PFAS soil to groundwater mass transfer is strongly influenced by the degree of flushing at these sites (see *e.g.*, Anderson et al. 2019). In other words, under low precipitation and deep groundwater, increased retention of PFAS is anticipated within the soil column (or decreased PFAS discharge to groundwater table) due to flushing limitations. Even though strong vadose zone retention is observed under the dry scenario, all three models are in excellent agreement on the simulated breakthrough curve peak magnitude and timing (Figure C-9). The maximum difference in peak magnitudes simulated by the three models for PFOA is 0.27% and for PFOS is 0.02%.

Including the effects of AWI under the dry scenario (**Figure C-10**), we see that the ANALYTICAL and HYDRUS models show excellent agreement for both soil types and constituents in terms of their magnitude and peak arrival time. As with the wet scenario, it is observed for both models that AWI decreases the magnitude of the peak concentrations and increases the arrival time of the peak at the water table for both PFOA and PFOS than when considering only linear, solid-phase sorption. The higher the interfacial adsorption coefficient at the AWI, the more pronounced the effects on peak concentrations and arrival times.

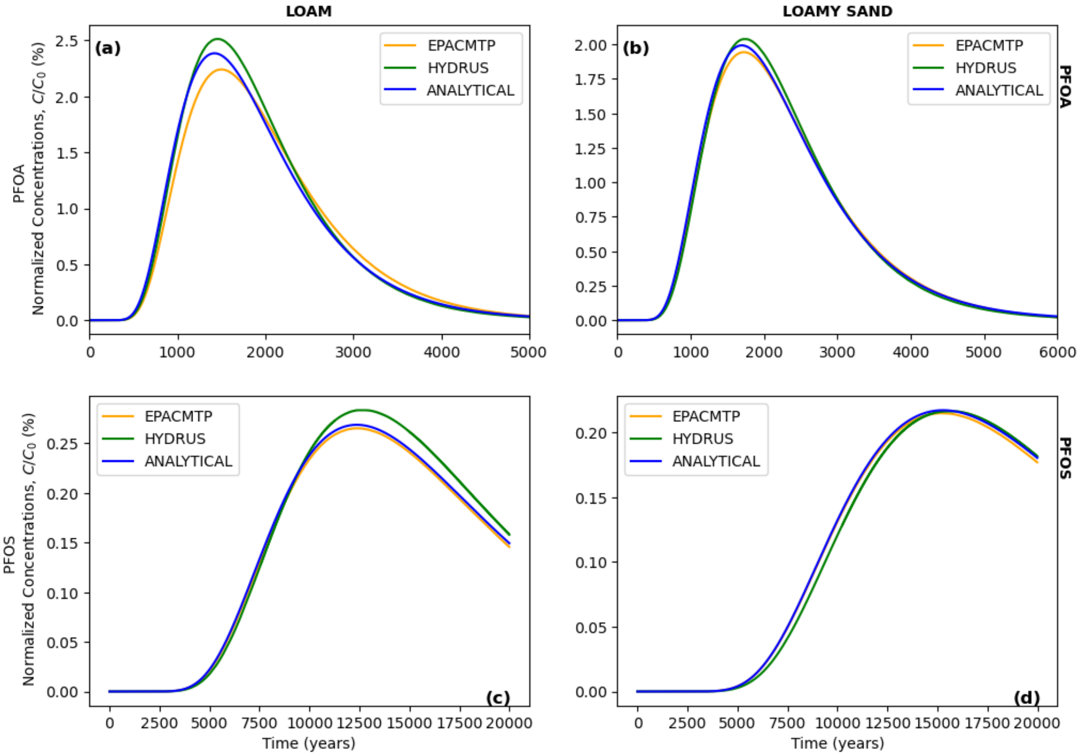


Figure C-9. Breakthrough curves at a 1-m water table depth in a dry using linear sorption.

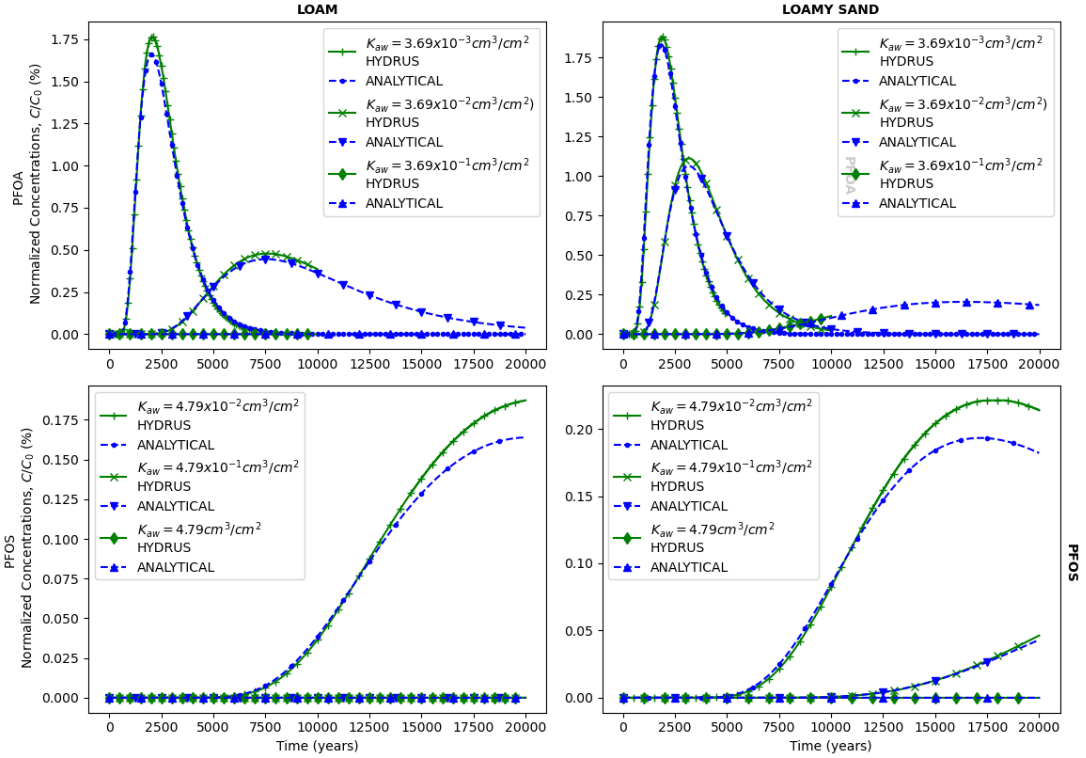


Figure C-10. Breakthrough curves at a 1-m water table depth in a dry environment using linear sorption and AWI effects for three values of Kaw (Silva et al., 2020).

Deeper Water Table, Wet Environment

For a 10 m soil column in a wet environment, the breakthrough curves show excellent agreement between the models for peak magnitude and arrival time for both constituents and soil textures under linear sorption only (**Figure C-11**) and including the effects of AWI (**Figure C-12**). The PFOA/PFOS concentrations at the water table for a 10 m soil column are much lower as compared to the 1 m soil column under the wet scenario. For example, under the wet scenario for a 1 m Loam soil column, approximately 90% of the input PFOA concentrations were observed at the water table (**Figure C-7**) while approximately 12% of the input PFOA concentrations were observed at the water table for the same conditions in the 10 m column. This is because the same total input mass is applied to the top of both soil columns, but the deeper soil column has larger soil volume and therefore greater sites for solid-phase sorption of the same input mass.

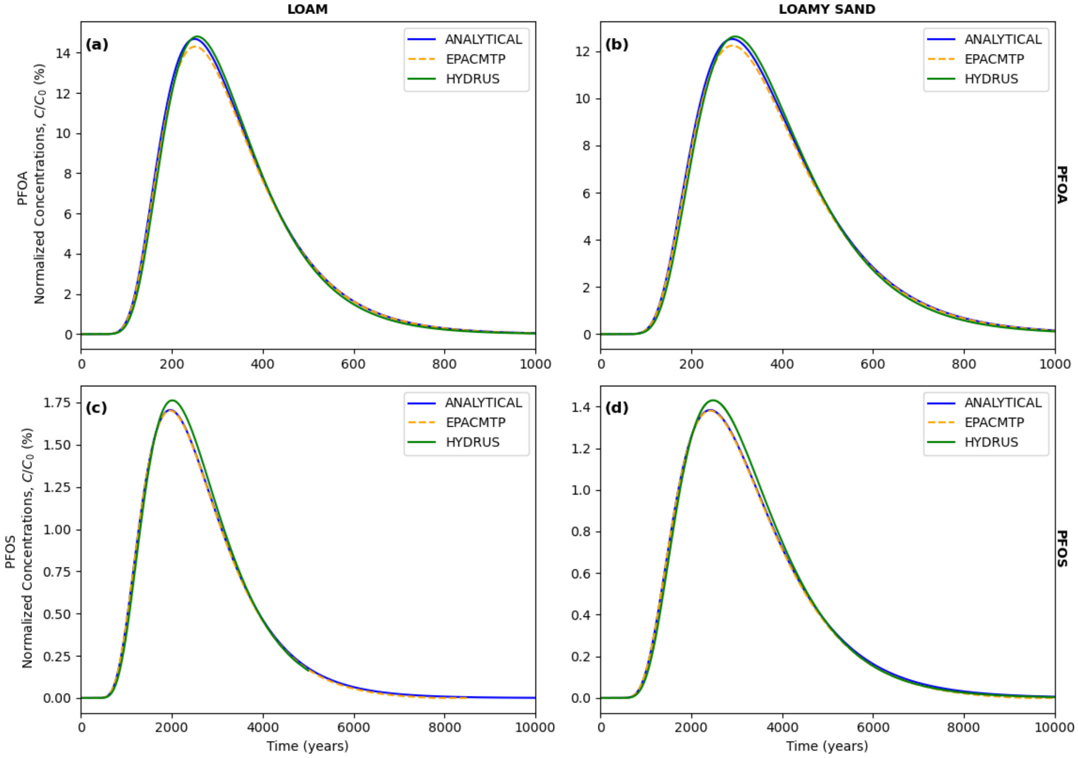


Figure C-11. **Breakthrough curves at a 10-m water table depth in a wet environment using linear sorption.**

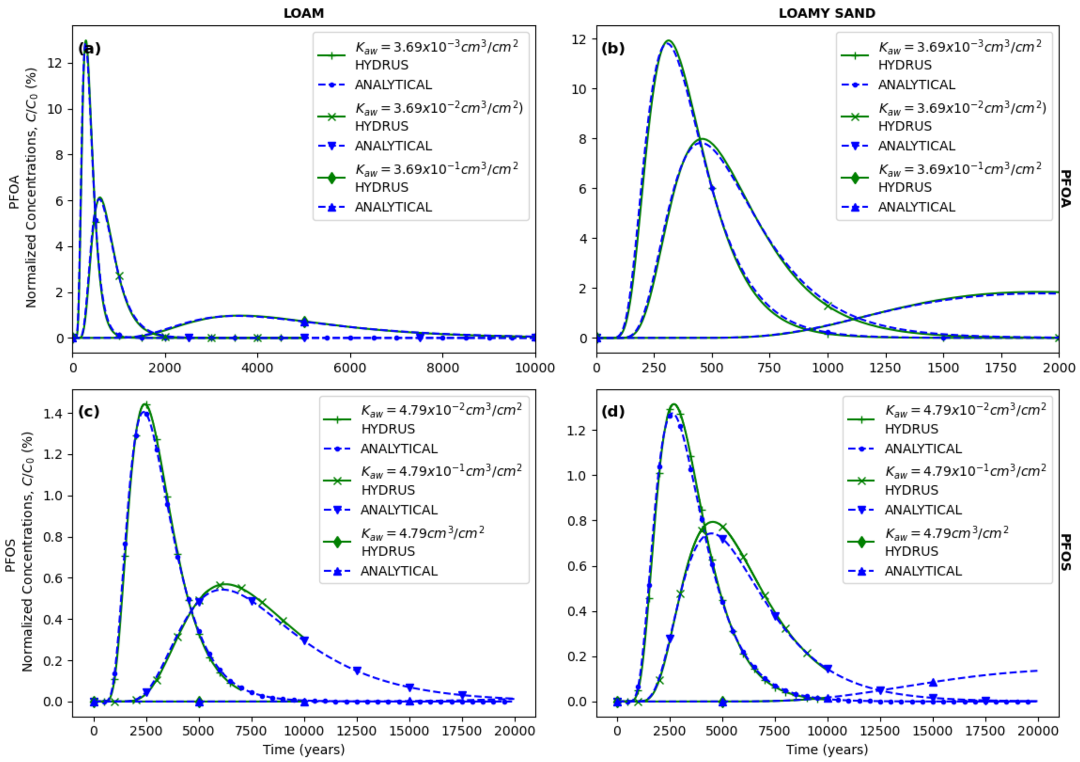


Figure C-12. Breakthrough curves at a 10-m water table depth in a wet environment using linear sorption and AWI effects for three values of Kaw (Silva et al., 2020).

Deeper Water Table, Dry Environment

For a 10 m soil column in a dry environment, the breakthrough curves for PFOA assuming solid phase adsorption only (**Figure C-13**, top row) correspond well between all models tested. Under the same scenario, all three models suggest that PFOS is not transported in the timeframe modeled to the water table due to strong solid phase adsorption. However, the simulated breakthrough curves are shown for completeness, but the reader will observe that the simulated concentrations are insignificantly low (<$8x10^{-7}$% of input concentrations; Figure C-13, bottom row). Finally, including the effects of AWI to PFOA/PFOS (**Figure C-14**, first column) shows good agreement between HYDRUS and ANALYTICAL model-simulated breakthrough curves. However, while the ANALYTICAL model was able to simulate the breakthrough curves for Loamy Sand under the same scenario, the numerical solution of HYDRUS became unstable beyond approximately 2,500 years for PFOA and 3,500 years for PFOS and the solutions did not converge. As noted earlier, the simulated PFOA/PFOS arrival time at the water table based on the breakthrough curves (Figure C-7) are not absolute and are only illustrative for model comparison purposes.

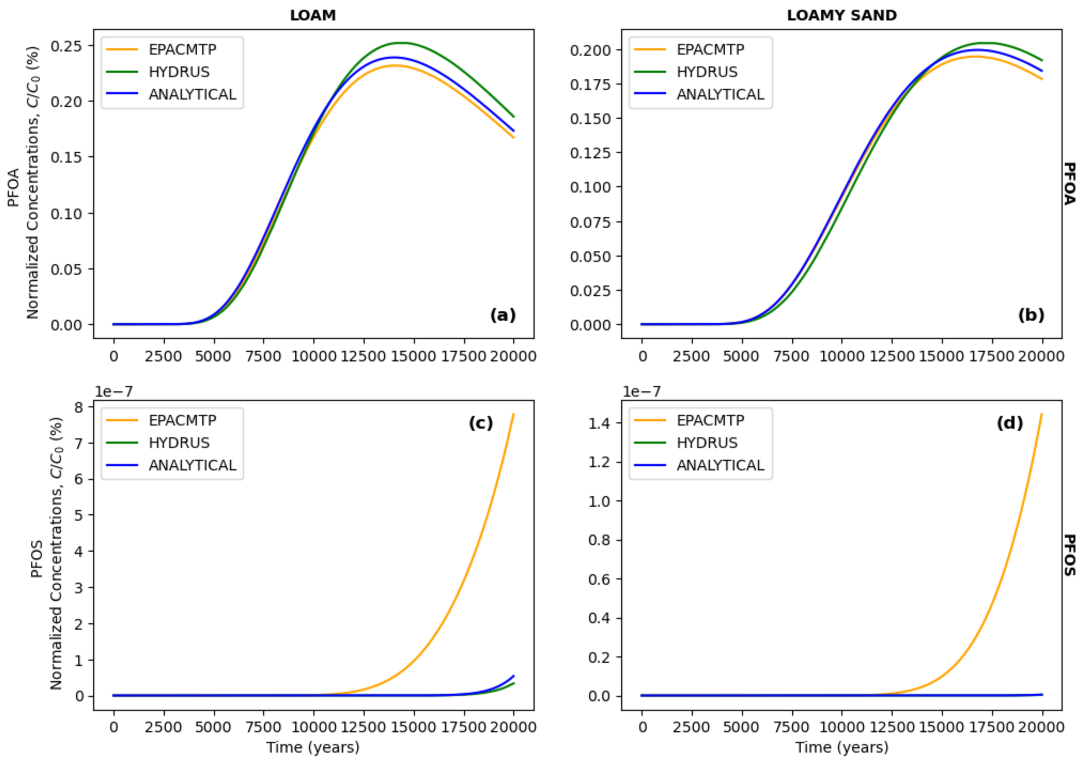


Figure C-13. Breakthrough curves at a 10-m water table depth in a dry environment using linear sorption.

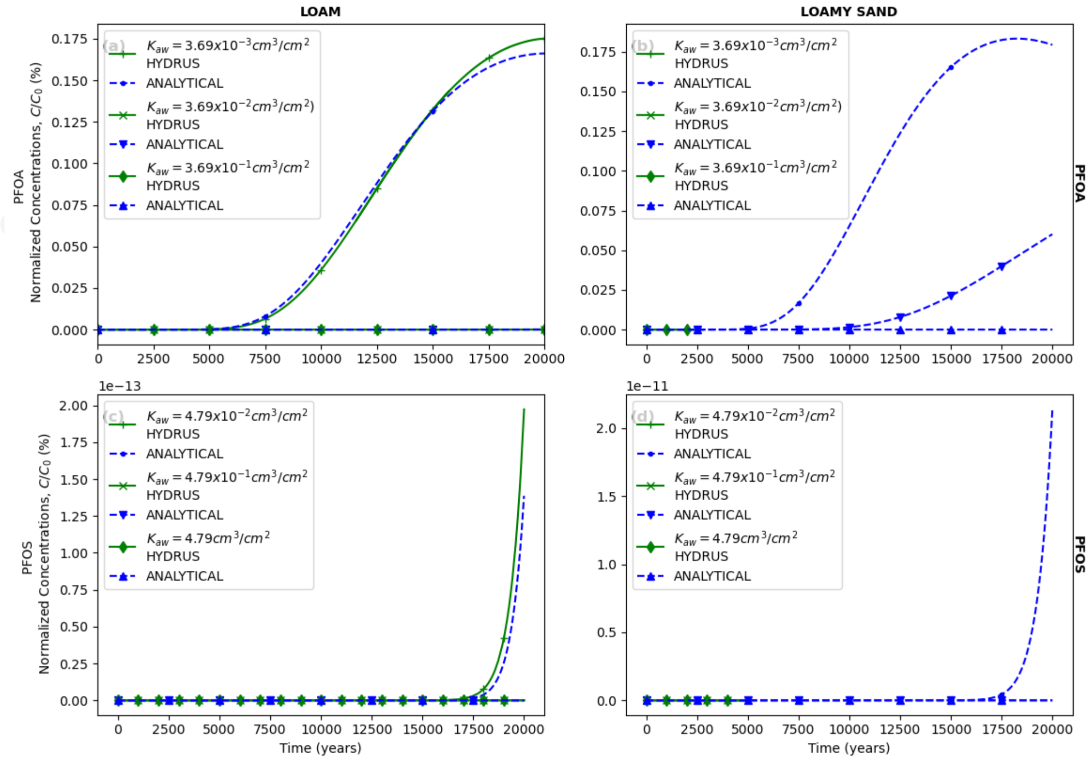


Figure C-14. Breakthrough curves at a 10-m water table depth in a dry environment using linear sorption and AWI effects for three values of Kaw (Silva et al., 2020).

C.1.5 Model Selection Conclusions

Three simulation models—EPACMTP, ANALYTICAL, and HYDRUS—were examined to determine which is best suited to support risk assessment objectives. The models were evaluated by comparing flow and PFOA/PFOS transport results from eight scenarios that reflect a broad range of key hydrogeologic conditions on a national scale including depth to water table, soil texture and meteorological conditions. Additionally, transport simulations included comparison of model outputs assuming solid phase adsorption only as well as solid phase adsorption with AWI effects.

Overall, the simulation results confirm that there is little difference between three models in simulating variable saturated flow regardless of soil textures, meteorological environments, and vadose zone thickness. Comparison of breakthrough curves at the water table when considering only linear, solid-phase sorption from transport simulations show excellent agreement between all three models for both soil columns (1m and 10m), soil types (loam and loamy sand), infiltration (dry and wet) and constituents (PFOA and PFOS) in terms of their magnitude and peak arrival time. The maximum difference in peak magnitudes for the various scenarios simulated by the three models for PFOA is less than 0.3% and for PFOS is 0.03%. However, the magnitude and peak arrival times were observed to be different between soil types, infiltration scenarios, constituent simulated and soil column depths. Our modeling results show that when biosolids are land applied at the surface, the greatest mass of PFAS arriving at the water table (~90% of input concentrations) in the shortest amount of time (~54 years) is observed for PFOA moving through a short, 1 m vadose zone under wet conditions. In contrast, all three models suggest that PFOS is not transported to the water table in a 10 m soil column in a dry environment due to strong solid phase adsorption. As noted earlier, the simulated PFOA/PFOS peak magnitudes and arrival time at the water table based on the breakthrough curves are not absolute or site-specific but only illustrative for model comparison purposes. Nevertheless, these bounding simulations highlight the importance of selecting appropriate values for location-specific and contaminant-specific critical factors such as K_d, K_{aw}, soil texture, depth to water table and net infiltration when conducting risk assessments, a conclusion that was also suggested by Pepper et al. (2023). Additionally, all solid phase adsorption simulations were performed assuming instantaneous equilibrium. However, the ANALYTICAL model is capable of simulating kinetics associated with solid phase adsorption, which maybe an important process in real soils with organic carbon or in clayey soils (see *e.g.*, Guelfo et al., 2020 and Schaefer et al., 2021).

Including the effects of AWI, we see that the ANALYTICAL and HYDRUS models show excellent agreement for both soil types and constituents in terms of their magnitude and peak arrival time. Generally, it is observed for both models that AWI decreases the magnitude of the peak concentrations and increases the arrival time of the peak at the water table for both PFOA and PFOS compared to models considering only linear, solid-phase sorption. The higher the interfacial adsorption coefficient at the AWI, the more pronounced the effects on peak concentrations and arrival times. This is consistent with increased retardation of PFOA/PFOS anticipated with retention at the AWI. While both models were able to simulate solid-phase and AWI retention processes and showed good agreement between model-simulated breakthrough curves, the numerical solution of HYDRUS became unstable for a 10 m soil column in a dry environment, while the ANALYTICAL model did not have any issues.

The contribution of PFOA/PFOS mass retention at the AWI was further evaluated at Boulder (Dry climate, deep water table) and Charleston (wet climate, shallow water table) using the ANALYTICAL model, K_{oc} bounds, and location-specific environmental parameters discussed in **Section C.3**. Model simulation results indicate that AWI retention is not a significant mechanism for PFOA/PFOS retention for the specific chemical- and environment-specific conditions modeled. Less than 1% of the total PFOA/PFOS mass leached from biosolids applied to the LAU is retained at the AWI with the remaining applied mass either sorbed to solid-phase or transported through aqueous phase under high-K_{oc}

conditions. Under low-K_{oc} conditions, the AWI contributes 7-9% of total leached PFOA/PFOS mass, except at Charleston for PFOS where 18% of total leached PFOS mass is retained at the AWI. These results suggest that PFOA/PFOS mass retained at the AWI is not a significant contributor to mass retention for the chemical- specific and environment-specific conditions modeled.

The overall objective of the preceding analysis was to evaluate transport processes available in unsaturated zone flow and transport simulators to predict PFOA/PFOS migration through the vadose zone to the water table for a range of environmental settings and constituent-specific fate and transport parameters. Overall, we observe that the vadose zone module in EPACMTP would produce higher (*i.e.*, risk-conservative) PFAS concentrations at the water table because the model does not have the ability to address PFAS-specific retention behavior at the AWI. While both HYDRUS and the ANALYTICAL models are capable of simulating PFAS-specific retention behavior, and generally in good agreement when simulating PFOA/PFOS leaching from surface soils resulting from the application of biosolids through the vadose zone to groundwater, these models require site-specific inputs to model AWI that are not available in the current risk assessment framework. Further, though the time to breakthrough on all models are longer than existing field studies indicate are possible, incorporating AWI into the HYDRUS and ANALYTICAL models only increases the time lag observed in the models compared to the monitored data. Evaluation of PFOA/PFOS mass retained at the AWI was not determined to be a significant contributor based on ANALYTICAL model simulations and the chemical-specific and environment-specific conditions discussed in **Section C.3**. Therefore, EPACMTP is used to conduct unsaturated and saturated zone flow and transport simulations to evaluate the fate and transport of PFOA and PFOS in land applied biosolids in this risk modeling framework.

C.2 Overview of EPACMTP

The transport of leachate from the land-applied biosolids or sewage sludge managed in surface disposal units through the unsaturated and saturated zones is evaluated quantitatively using EPACMTP (US EPA, 2003a,b,d; 1997). EPACMTP simulates the flow and transport of contaminants in the unsaturated zone and aquifer beneath a waste management unit to yield the concentration that arrives at a specified receptor location. The LAU and SDU source models determine the leachate concentration used as an input to EPACMTP. As described in the *Addendum to the EPACMTP Technical Background Document* (US EPA, 2003a), new functionality was added to the EPACMTP model to create a dynamic, mass-conserving linkage between the source models and EPACMTP.

The groundwater model accounts for advection, hydrodynamic dispersion, equilibrium linear or nonlinear sorption, and transformation processes via chemical hydrolysis. In this analysis, data were compiled from the scientific literature to develop organic carbon partition coefficients to simulate equilibrium linear partitioning for PFOA and PFOS. Organic carbon partition coefficient inputs are discussed in **Section C.3** and **Appendix B**.

EPACMTP is a composite model that consists of two coupled modules: (1) a 1-dimensional (1-D) module that simulates vertical infiltration and dissolved contaminant transport through the unsaturated zone, and (2) a saturated zone flow and transport module that includes three groundwater transport solution options: (i) fully 3-D transport, (ii) quasi-3-D transport (a combination of cross-sectional and areal solutions), and (iii) pseudo-3-D transport (hybrid analytical and numerical solution). The applicability and appropriateness of each of the transport solution options depend on the problem considered. The pseudo-3-D solution is the most computationally efficient of the available options. In addition, the pseudo-3-D solution can accurately and efficiently generate full breakthrough curves at the receptor location. For these reasons, the pseudo-3-D solution option was chosen for this analysis.

The EPACMTP algorithms assume that the soil and aquifer are uniform porous media and that flow and transport are described by Darcy's law and the advection-dispersion equation, respectively. EPACMTP does not account for preferential pathways, such as fractures and macropores, or facilitated transport, which may affect the migration of PFAS. For example, Zeng and Guo (2021) have shown that preferential flow pathways generated by soil heterogeneities can reduce the strength of retention at the air-water interfaces that can lead to early arrival and accelerated leaching of (especially long-chain) PFAS.

EPACMTP models the advective movement in the unsaturated zone as 1-D, whereas the saturated zone module accounts for 3-D flow and transport. EPACMTP also considers mixing due to hydrodynamic dispersion in both the unsaturated and saturated zones. In the unsaturated zone, flow is gravity-driven and prevails in the vertically downward direction. Therefore, the flow is modeled in the unsaturated zone as 1-D in the vertical direction. It is also assumed that transverse dispersion (both mechanical dispersion and molecular diffusion) is negligible in the unsaturated zone. This assumption is reasonable given that lateral migration due to transverse dispersion is negligible compared with the horizontal dimensions of the waste management unit. In addition, this assumption is environmentally protective because it allows the leading front of the pollutant plume to arrive at the water table with greater peak concentration in the case where the duration of leaching is finite.

In the saturated zone, the movement of pollutants is primarily driven by ambient groundwater flow, which in turn is controlled by a regional hydraulic gradient and hydraulic conductivity in the aquifer formation. The model considers the effects of infiltration from the waste source and the regional recharge into the aquifer. The effect of infiltration from the waste source is an increase in groundwater flow in the horizontal transverse and vertical directions underneath and in the immediate vicinity of the waste source, as may result from groundwater mounding. This 3-D flow pattern will enhance the horizontal and vertical spreading of the plume. Regional recharge outside of the waste source causes a (vertically) downward movement of the plume as it travels in the (longitudinally) downgradient groundwater flow direction. In addition to advective movement and groundwater flow, the model simulates the mixing of contaminants with groundwater due to hydrodynamic dispersion, which acts in the longitudinal direction (*i.e.*, along the groundwater flow direction) and in the horizontal and vertical transverse directions.

Leachate pollutants can be subject to complex geochemical interactions in soil and groundwater, which can strongly affect their rate of transport in the subsurface. EPACMTP treats these interactions as equilibrium-sorption processes. The equilibrium assumption means that the sorption process occurs instantaneously, or at least very quickly, relative to the time scale of pollutant transport. However, studies have observed that PFOA/PFOS solid phase sorption processes are not always well represented by reversable equilibrium partitioning assumptions due to rate-limited sorption considerations (Guelfo et al., 2020; Brusseau, 2020). Guo et al. (2022) implemented a linear isotherm simplification for solid phase adsorption and compared predicted simulations for a wide range of sand-packed miscible-displacement experiments for PFAS under water-unsaturated conditions as well as a simulation of PFAS leaching at a model AFFF-impacted fire training area site. These authors found that their model with linear isotherm simplification reproduced solutions identical to a full-scale numeric model that accounts for a set of comprehensive PFAS-specific transport processes, including nonlinear solid phase adsorption. While this is an active area of research, uncertainties in PFAS-specific, non-linear behavior in assessing the exposures associated with land-applied biosolids on the groundwater pathway may need consideration. Although sorption, or the attachment of leachate pollutants to solid soil or aquifer particles, may result from multiple chemical processes, EPACMTP combines these processes into an effective soil-water partition coefficient (K_d). The retardation factor, R, accounts for the effects of equilibrium sorption of dissolved pollutants onto the solid phase. *R*, a function of the pollutant-specific K_d and the soil or aquifer properties, is calculated as follows:

$$R = 1 + \frac{\rho_b K_d}{\phi}$$

where

R = retardation factor (unitless)
ρ_b = soil or aquifer bulk density (g/cm^3)
K_d = solid-water partition coefficient (cm^3/g)
ϕ = water content (in unsaturated zone) or porosity (in saturated zone) (unitless).

Chemicals with low K_d values will have low retardation factors, which means that they will move at nearly the same velocity as the groundwater. Chemicals with high K_d values will have high retardation factors and may move many times slower than groundwater.

As modeled in EPACMTP, the K_d of an organic pollutant is assumed to be constant within each modeled soil column and is calculated as the product of the mass fraction of organic carbon in the soil or aquifer and a pollutant-specific organic carbon partition coefficient (K_{oc}). Multiple literature searches were conducted to identify field and laboratory studies reporting either measured or estimated values of K_d or K_{oc}, with and without associations to biosolids or land application of biosolids, for surface and subsurface soils, aquifer materials, and settled and suspended surface water sediments. The results of the literature survey, discussed in **Section C.3**, demonstrated that the spread and median values of log K_{oc} tend to show less variability across the various categories of field and laboratory studies, biosolids and non-biosolids related studies, and across media than log K_d. This behavior is reasonable as log K_d additionally reflects the effects of organic carbon (OC) variability in various matrices, as well as other parameters (*e.g.*, pH), whereas log K_{oc} does not. Therefore, K_{oc} was used as inputs to the model, along with fraction of organic carbon corresponding to the dominant soil mega texture at each location (**Section C.3**), model sensitivity tested at a lower and upper bound K_{oc} values as described in **Appendix D**.

EPACMTP simulates steady-state flow in both the unsaturated and saturated zones and can accommodate either steady-state or transient contaminant transport. Steady-state transport modeling is a protective modeling approach in which a unit continues to release contaminants indefinitely (continuous source); eventually, the model will predict that the receptor well concentration reaches a constant value. However, in this analysis, transient transport simulations were performed. This finite source approach simulates the amount of time over which the land application unit is active and the time-dependent movement of chemical pollutants in the subsurface to the receptor well.

C.2.1 Groundwater Receptor

One of the most important inputs for EPACMTP is receptor location, which for this risk analysis included a residential drinking water well located 5 meters from the edge of the farm field (*i.e.*, center of the buffer). EPACMTP can also evaluate the exposure concentration of a hypothetical residential drinking water well at a specified depth below the water table. For this evaluation, four depths below the water table were considered (0.5 m, 1.0 m, 1.5 m and 2.0 m) and the maximum of the peak concentrations at all depths was considered as the exposure concentration. The well depths were limited to the top 2.0 m below the water table (1) to be consistent with a residential well scenario (these wells are generally shallow because of the higher cost of drilling a deeper well) and (2) to produce a conservative estimate of risk (because the infiltration rate is generally lower than the groundwater seepage velocity, groundwater plumes tend to be relatively shallow). Limitation of well depth is further validated by the consistent PFOA and PFOS groundwater concentration profiles with depth modeled at the residential drinking water well located 5 meters from the edge of the farm field (**Figures C-15** and **C-16**). These profiles show that contamination is roughly constant over the top 6-8 m of the aquifer at all modeled site locations at the residential drinking water well and alleviates concerns on overpredicting modeled risks by selecting the maximum of the peak concentrations within the top 2.0 m below the water table.

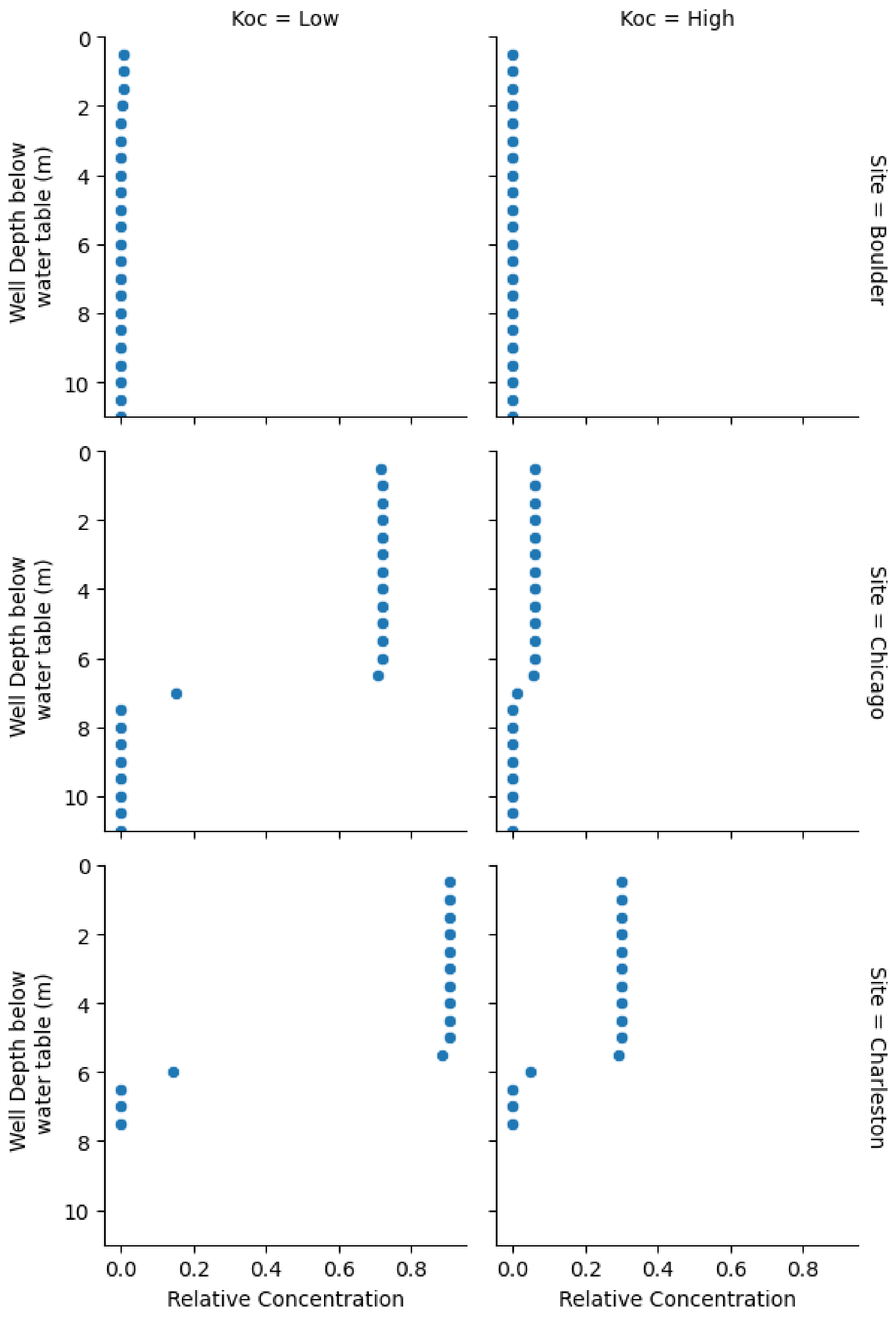


Figure C-15. Well depth below water table (m) vs. relative PFOA concentrations *for point observations (solid circles) for CROP, low Koc (left panels) and high Koc (right panels) at a well located 5 meters away from edge of field.*

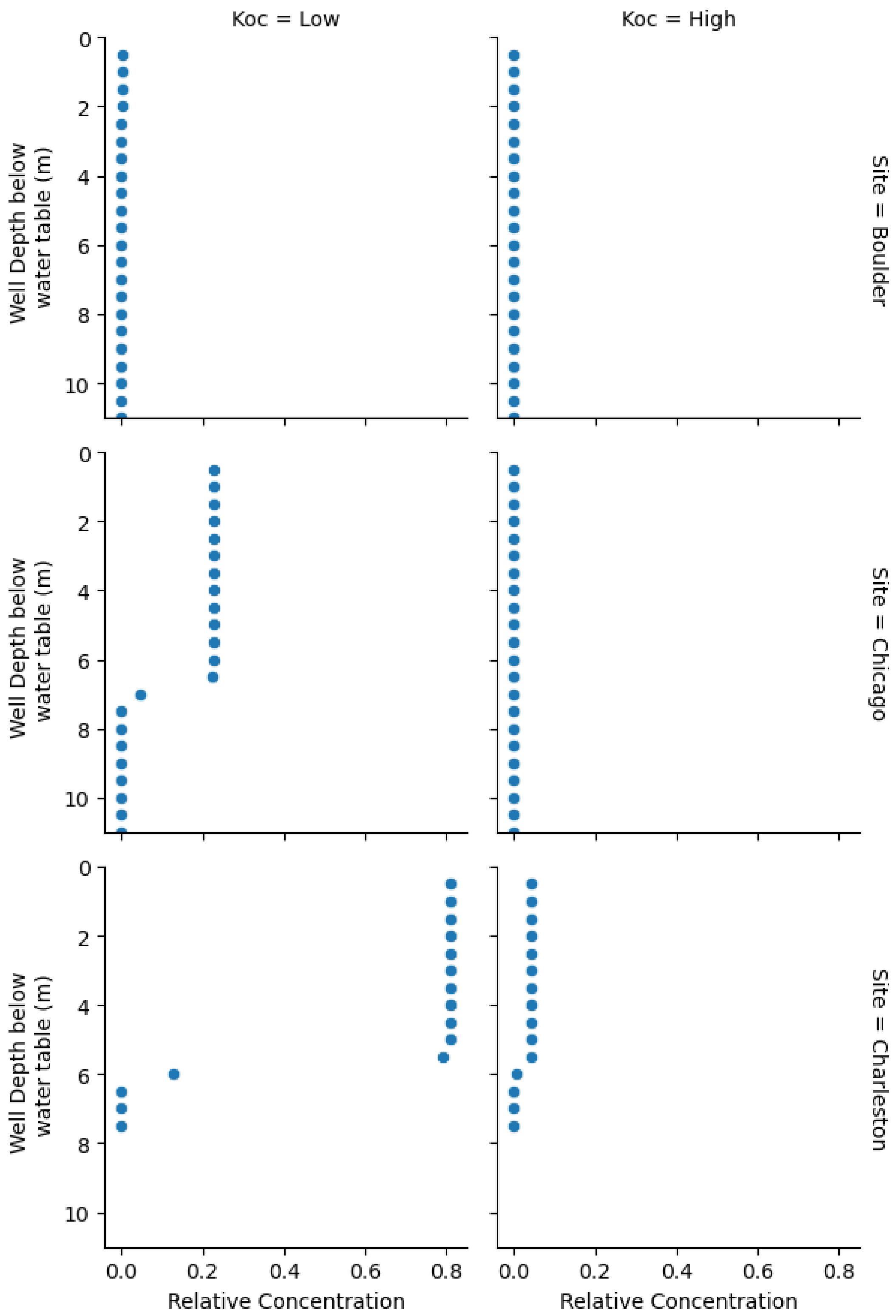


Figure C-16. Well depth below water table (m) vs. relative PFOS concentrations *for point observations (solid circles) for CROP, low Koc (left panels) and high Koc (right panels) at a well located 5 meters away from edge of field.*

C.2.2 Groundwater Pathway Simulations

The leachate fluxes (g/m^2/yr) and infiltration water fluxes (m/d) estimated by the land application and surface disposal source models were used as input to the groundwater fate and transport model, EPACMTP, to generate pollutant concentrations at receptor wells. These fluxes serve as the flow and transport boundary conditions within the footprints of the field and disposal units, The leachate fluxes (g/m^2/yr) and infiltration water fluxes (m/d) were estimated using regional and local variables at three geographic regions (approximated using data from Boulder, Chicago and Charleston, USA) corresponding to dry, moderate and wet climate conditions. For the groundwater pathway simulations in this analysis, the flux of pollutants from the LAU and the SDU were not constant. Instead, the source models predict a time series of leachate flux, whereby the mass transfer to the groundwater pathway varies from year to year. The maximum leachate flux and corresponding annual infiltration rate from the profiles generated by the source models were used as inputs to EPACMTP along with various chemical-, location-, and environment-specific variables, as discussed in **Section C.3** below.

C.2.3 Key Assumptions

This section presents key assumptions associated with the groundwater modeling approach. More comprehensive documentation of EPACMTP and associated assumptions are available in the *EPACMTP Technical Background Document* (US EPA, 2003a).

- The model assumes that the vertical migration is 1-D and that transverse dispersion is negligible in the unsaturated zone.
- The model assumes linear equilibrium sorption for PFOA/PFOS in the unsaturated soil and aquifer zones and homogeneous aquifer conditions.
- The model assumes that receptors use the uppermost (water table) aquifer, rather than a deeper aquifer, as a source of drinking water. This assumption could overestimate risks in cases in which the uppermost aquifer is not used.
- The model assumes that long-term average conditions are sufficient for exposure calculation and that shorter frequency fluctuations (*e.g.*, in rainfall/infiltration) are insignificant in estimating long-term risk.
- Biodegradation in groundwater was excluded given the recalcitrant nature of PFOA/PFOS.
- Preferential flow in karst aquifers or in fractures was not considered, although such conditions are known to exist over broad areas. Preferential flow can allow contamination to migrate faster and at a higher concentration than in a standard porous medium. However, the contamination typically does not spread over such a broad area. As a result, the modeling may underestimate or overestimate the concentrations in groundwater, depending on how concentrations are averaged spatially and temporally.

C.3 Model Inputs

Appendix B presents the input values used in modeling the groundwater pathway using EPACMTP. Below is a description of some key EPACMTP inputs.

C.3.1 Fluxes from Source Models

The releases of PFOA/PFOS mass and infiltrating water were determined using waste management unit-specific models (land application unit, or LAU, and surface disposal unit, or SDU) developed for 3MRA. These models generate time-series of PFOA/PFOS mass fluxes and infiltrating water fluxes to the subsurface as well as releases to other exposure pathways, the latter a capability not available in the source terms provided in EPACMTP. Therefore, to satisfy the multi-pathway analysis plan for this risk assessment, the 3MRA waste management unit models are used provide mass and water fluxes to EPACMTP for fate and transport simulations of the subsurface environment. For example, **Figure C-17**

(a) through (c) shows the time series of PFOA and PFOS mass fluxes and infiltrating water fluxes leaching to the subsurface as generated for the Pasture LAU scenario in Chicago (moderate meteorological conditions). PFOA and PFOS concentrations in the infiltrating water resulting from the corresponding mass and infiltrating water fluxes for PFOA and PFOS, assuming a "low-K_{oc}" value (discussed in **Section C.3**) are shown in **Figure C-17** (d) and (e).

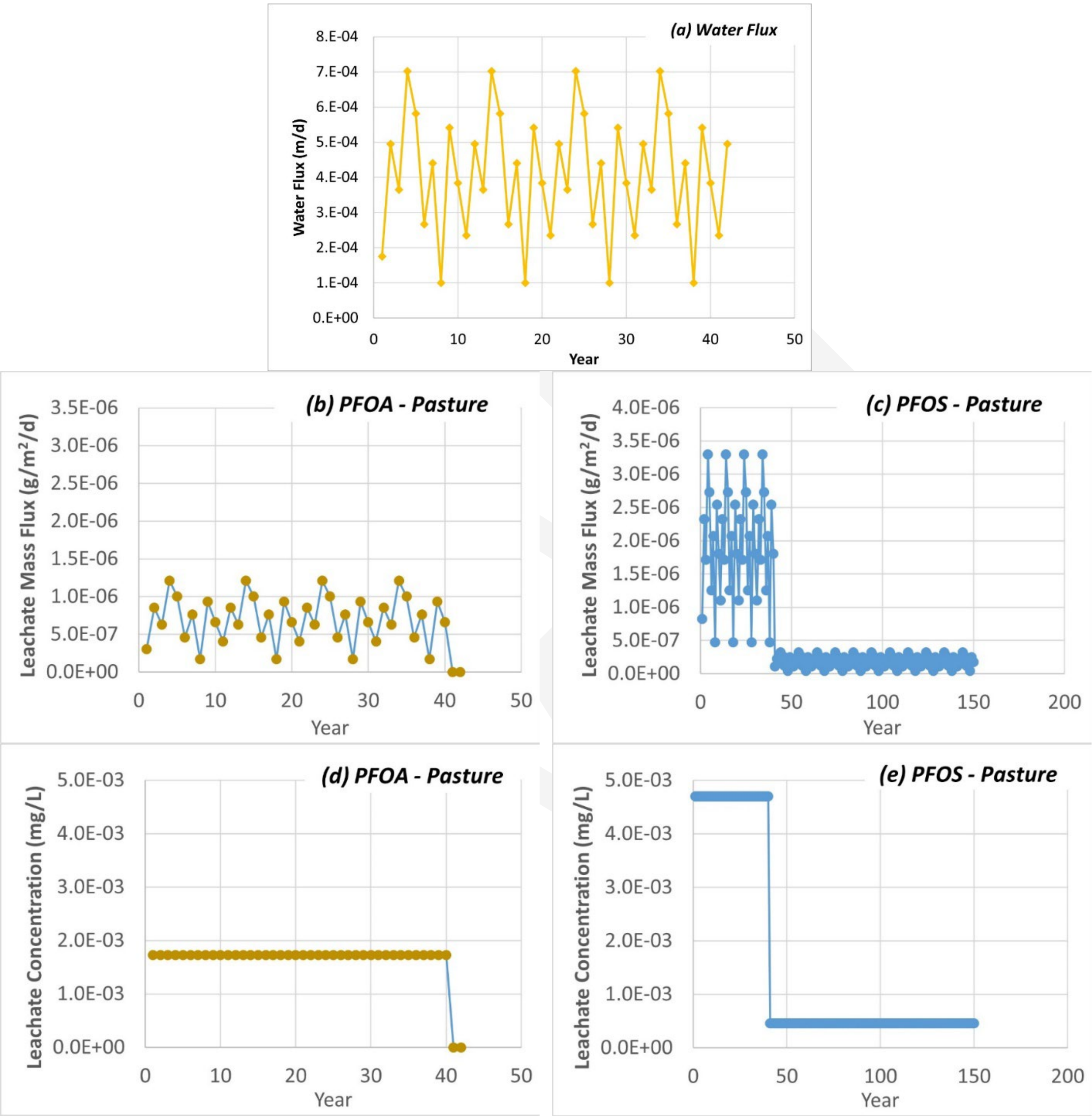


Figure C-17. Simulated time series of PFOA and PFOS mass and water fluxes generated for the pasture LAU scenario in Chicago, representing moderate meteorological conditions. (a) infiltrating water fluxes; (b) and (c) PFOA and PFOS mass fluxes leaching to the subsurface; and (d) and (e) PFOA and PFOS concentrations in the infiltrating water resulting from the corresponding mass and infiltrating water fluxes.

In Figure C-17 (a), the simulated water fluxes at Chicago, representing moderate meteorological conditions, vary between $5.5\text{x}10^{-5}$ m/d to $4.2\ \text{x}10^{-4}$ m/d. EPACMTP simulations presented in this report assume a uniform water flux corresponding to the maxmimum leachate flux value at each location and source model, as summarized in **Appendix B**. Note that the water fluxes only vary by source model and geographic location and independent of chemical constituent simulated.

Similar to the water fluxes, the leachate fluxes (**Figure C-17**(b) and (c)) entering the subsurface vary from year to year for PFOA and PFOS. For the Pasture LAU scenario, leachate concentration and flux variations are greatest for the first 40 years, the modeled duration of leaching from the source in the pasture LAU scenario.

Maximum leachate fluxes (g/m^2/d) estimated by the land application and surface disposal source models were used as input to EPACMTP to generate PFOA and PFOS concentrations. PFOA and PFOS concentrations (**Figure C-17**(d) and (e)) in the infiltrating water remains constant during the modeled duration of leaching from the source (*e.g.*, 40 years for pasture LAU scenario). All leachate fluxes from the source models were applied uniformly over the footprint of the either the LAU or SDU at the top of the unsaturated soil column. **Appendix B** summarizes the input maximum leachate PFOA and PFOS concentrations for various source models and geographic locations.

C.3.2 Koc

The primary chemical-specific input parameters of concern within the groundwater pathway for PFOA and PFOS are their organic carbon distribution coefficient (K_{oc}) and effective diffusion coefficient in water (D*). Under natural soil-water conditions, volatilization of PFOA and PFOS is negligible (Johansson et al., 2017; Sima and Jaffé 2021), making inputs like diffusion coefficient in air and Henry's law constant irrelevant. PFOA and PFOS also do not degrade, so degradation rates are also not relevant. Values for D* are straightforward and provided in Appendix B (Table B-1). Koc, however, is highly variable and this section describes the literature review conducted to establish input values for EPACMTP modeling.

A review of measured K_{oc} and solid phase adsorption coefficient (K_d) values reported in literature for PFOA and PFOS was conducted with and without associations to biosolids or land application of biosolids, for surface and subsurface soils, aquifer materials, and settled and suspended surface water sediments. This section describes the literature search methodology, data selection and review, data extraction, results and conclusions.

C.3.2.1 Literature Search Methodology

The EPA conducted multiple literature searches, the most recent of which was conducted on March 18, 2024, to identify papers addressing per- and polyfluoroalkyl substances (PFAS) in general as well as PFOA and PFOS specifically. Data were analyzed and categorized employing well defined data quality criteria, summarized, and finally evaluated for use in modeling exercises.

This section describes the overall search methodology, including the databases searched, specific search strings, and the abstract review strategy and article selection.

The following online databases were searched:

- PubMed
- Web of Science (includes Science Citation Index Expanded, Social Sciences Citation Index, and Conference Proceedings Citation Indexes for Science and for Social Science and Humanities)
- Environment Complete
- CAB Abstracts
- Fish, Fisheries & Aquatic Biodiversity Worldwide
- TOXLINE

- ScienceDirect.

Results were restricted to papers in English published after 1990 up through to the date of the last literature search.

Three sets of searches were conducted: (a) a broad search that did not specify a relationship to biosolids and included per- and polyfluoroalkyl substances (PFAS) in general; (b) a search that looked specifically for PFOA or PFOS and for mention of biosolids or land application and other related terms; and (c) a search focused solely on solid-phase partitioning and sorption of PFOA and PFOS. Searches (a) and (b) were broader than just solid-phase sorption coefficients.

Broad search (a) included PFOA or PFOS as well as additional PFAS. These search strings were formulated in the three following parts:

Part 1:
("perfluoroalkyl substance*" OR "polyfluoroalkyl substance*" OR "PFAS" OR "PFASs" OR "PFOA" OR "perfluorooctanoic acid" OR "PFOS" OR "perfluorooctane sulfonic acid" OR "perfluorooctanesulfonic acid" OR “PFNA” OR “perfluorononanoic acid” OR “heptadecafluorononanoic acid” OR “perfluoro-n-nonanoic acid “OR “PFHxS” OR “perfluorohexanesulfonic acid” OR “perfluorohexane-1-sulphonic acid” OR "Gen X" OR "GenX" OR "stain repellent*" OR "water resistant" OR "aqueous film-forming foam" OR "AFFF" OR "perfluoroalkyl acid" OR "PFAA" OR "surfactant")

Part 2:
AND ("fate" OR "transport")

AND (“Retention” OR “Model”)

AND (“Adsorption” OR “Sorption”)

AND ((“Field” OR “Lab” OR “Laboratory”) AND (“Data” OR “Experiment”))

Part 3:
AND (“Vadose” OR “unsaturated”)

AND (“leaching").

Eight individual searches were conducted that used Part 1 plus each possible pair of the separate lines in Parts 2 and 3.

Broad search (b) targeted just PFOA and PFOS and was limited to papers published in 2017 or later. These search strings were formulated in four parts:

Part 1 – Constituents of Concern:
("PFOA" OR "perfluorooctanoic acid" OR "PFOS" OR "perfluorooctane sulfonic acid" OR "perfluorooctanesulfonic acid")

Part 2 – Properties:
AND ("uptake" OR "*transfer" OR “*accumulation” OR “BCF” OR “BAF” OR “propert*” OR “health” OR “effect” OR “diffusiv*” OR “partition*”)

Part 3 – Biosolids:
AND (“sewage sludge” OR “biosolids” OR “treated sewage” OR “sludge treatment” OR “sewage treatment”)

Part 4 – Land Application:
AND (“land application” OR “farm” OR “agriculture” OR “soil”)

Three individual searches were conducted using the following string combinations:

Search 1: Part 1 + Part 2 (to capture the universe of PFAS and properties).

Search 2: Part 1 + Part 3 (to capture the universe of PFAS and biosolids).

Search 3: Part 1 + Part 4 (to capture the universe of PFAS and land application).

Focused search (c) targeted keywords associated with solid-phase sorption for PFOA, PFOS, and PFAS, without limitations on publishing date. These search strings were formulated in two parts:

Part 1 – Constituents of Concern:
("PFOA" OR "perfluorooctanoic acid" OR "PFOS" OR "perfluorooctane sulfonic acid" OR "perfluorooctanesulfonic acid" OR “PFAS”)

Part 2 – Properties:
AND ("Koc” OR "partitioning coefficient" OR "organic carbon?water partitioning coefficient" OR “Kd” OR “soil *sorption coefficient” OR “*sorption coefficient” OR “distribution coefficient” OR “solid?liquid partitioning coefficient” OR “soil?water partitioning coefficient” ” OR “*sorption”)

One search was conducted using the following string combinations:

Search 1: Part 1 + Part 2 (capture the universe of PFOA and PFOS solid-phase sorption and properties).

C.3.2.2 Review and Data Selection Process

The results of the above searches were compiled, and duplicates removed, yielding 1,864 unique articles. We added two additional sources to those: Articles cited in the *PFAS Technical and Regulatory Guidance Document and Fact Sheets* (ITRC, 2022; Table 4.1) and data from the United States Geological Survey (USGS) for New Hampshire (Tokranov et al., 2023).

The EPA reviewed the abstracts, or if no abstract was available, the titles, and categorized them for further review for a variety of purposes based on keywords. For this review, we identified 234 articles that mentioned K_d or K_{oc} and PFOA or PFOS. We obtained the full text of those papers.

Upon reviewing the full text and evaluating the data quality, The EPA classified the articles into three types, based on whether they contained biosolids-related keywords (biosolid, wastewater treatment plant, or sewage sludge):

Studies in biosolids (46 articles)

Studies not in biosolids (169 articles)

Type could not be determined due to lack of clarity (19 articles; the EPA evaluated these further in the data extraction step).

C.3.2.3 Data Extraction

The EPA searched each article for partitioning data (K_d and/or K_{oc}) and identified whether the data were from field or laboratory experiments:

Field experiments included cases where the partitioning data was estimated directly from the original field sample condition or from the field sample spiked with PFOA/PFOS for concentration measurement purposes.

Laboratory experiments included cases where the partitioning coefficient was estimated through sorption/desorption or column experiments.

The EPA then assigned data to four categories:

Type A: biosolids-related field data

Type B: biosolids-related lab data

Type C: not biosolids field data

Type D: not biosolids lab data

C.3.2.4 Search Results

Table C-8 summarizes the data extracted from the literature review articles and other data sources. In total, the EPA extracted about 2,000 data points from 101 articles.

Table C-8. Summary of Data Extracted from Articles Identified in the Literature Search and USGS

Constituent	Scope	General Literature Review				NH/USGS Data[a]	
		log Kd		log Koc		log Kd	
		Field Studies	Lab Studies	Field Studies	Lab Studies	Field Studies	Lab Studies
PFOA	Biosolids	3	27	0	3	30	89
	Not biosolids	152	231	71	200	18	133
PFOS	Biosolids	0	14	0	11	22	86
	Not Biosolids	152	307	96	234	18	133

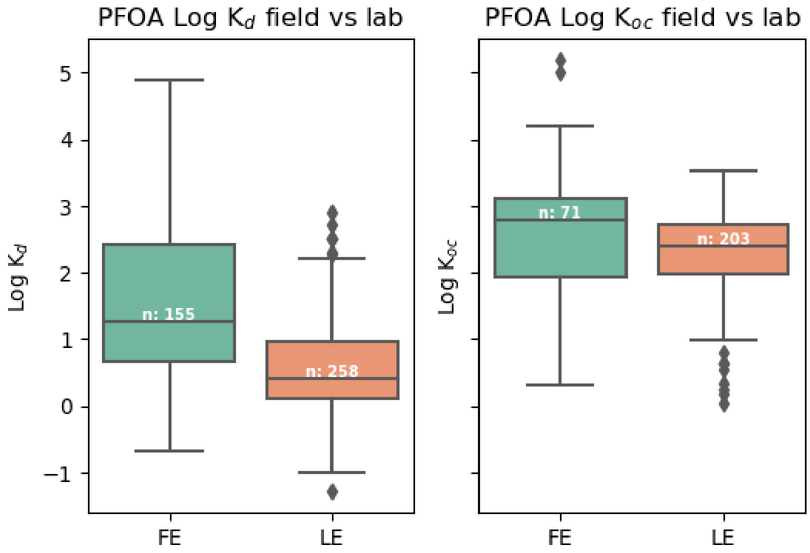
[a] Data reported by Tokranov et al. (2023).

The detailed results are presented by constituent below. The figures include only the literature search data unless otherwise specified. All values are presented as log K_d or log K_{oc}. Note that K_d is defined as the concentration in the solid phase divided by the concentration in the aqueous phase. Accordingly, a negative log K_d (*i.e.*, K_d less than 1) means that less constituent is present in the solid phase than the aqueous phase, and thus there is low solid phase sorption.

Results for PFOA

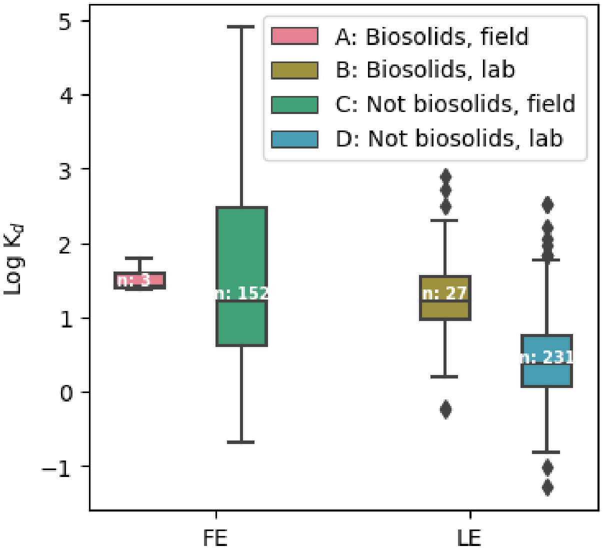
A significant finding for the purposes of identifying representative values of these parameters for modeling is that reported log K_d and log K_{oc} values span more than four orders of magnitude (**Figure C-18**). For log K_d, the median value from field data is greater than the median for laboratory data for PFOA (this is including both biosolids and non biosolids studies). Somewhat more than half of the PFOA data were from laboratory data (n = 258 for lab data, n = 155 for field data). Similarly, the median log K_{oc} for field data is greater than the median for laboratory data. Unlike K_d studies, however, approximately three times as many K_{oc} observations are from laboratory studies (n = 203) than are from field studies (n = 71), and no biosolids-related field data were identified.

The range of log K_d values for field studies not related to biosolids (Type C) is larger than the corresponding study type related to biosolids (Type A), and the range of the biosolids-related values is entirely encompassed within the range of not biosolids related values (**Figure C-19**). This may be due to the smaller number of biosolids-related field studies (3 reported values) than non-biosolids related field studies (152 reported values). However, the range of log K_d values are similar for lab studies related to biosolids (Type B) and not related to biosolids (Type D). The similar range of log K_d values for lab-studies maybe related to the larger pool of lab studies (258 reported values) than field studies (155 reported values). Note there were no biosolids-related log K_{oc} data for PFOA, so no equivalent comparison to the one shown in Figure C-19 for log K_d could be made.



Shaded portion represents the interquartile range (IQR); whisker ends are quartile ±1.5xIQR; diamonds are outliers.
FE=Field, LE=lab

Figure C-18. Boxplots of log K_d (left) and log K_{oc} (right) values for PFOA: field (FE) vs. laboratory (LE) studies.



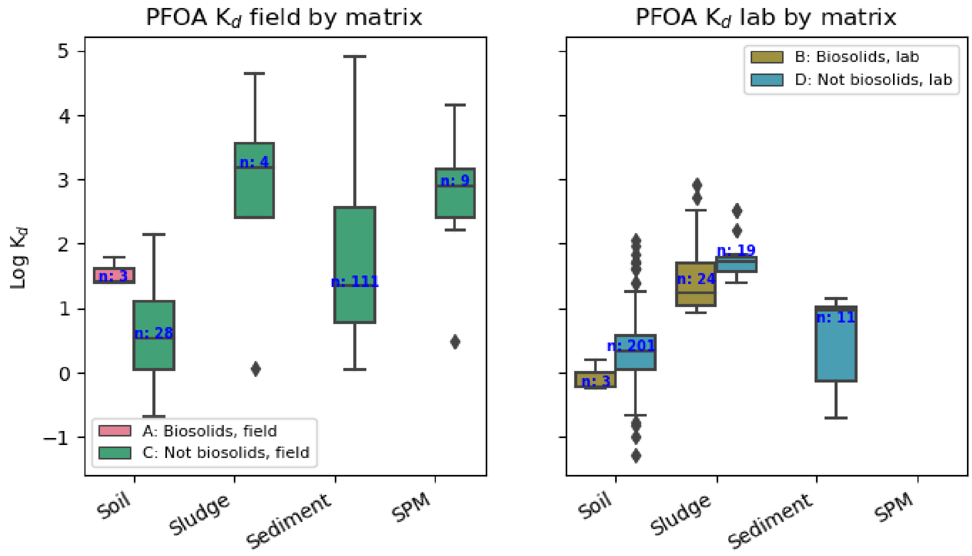
Shaded portion represents the interquartile range (IQR); whisker ends are quartile ±1.5xIQR; diamonds are outliers.
FE=Field, LE=lab

Figure C-19. Boxplot of log K_d values for PFOA: biosolids vs. not biosolids.

We further evaluated the data by sample media: soil, sludge, sediment, or suspended particulate matter (SPM). Most log K_d and log K_{oc} studies have been performed in soil (n=235 for K_d, n=204 for K_{oc}) or sediment (n=122 for K_d, n=62 for K_{oc}), with a lesser number in sludge (n=47 for K_d, n=4 for K_{oc}), and very few in SPM (n=2 for K_d, n=4 for K_{oc}).

The overall range of log K_d values in soil from field and lab studies spans approximately three orders of magnitude (log K_d approximately –1.3 to 2.1; **Figure C-20**). However, this is clearly dominated by non-biosolids related lab studies for both media (green and blue bars for Types C and D in Figure C-20). The range for biosolids-related values for soil (there are none for sediment) cover a considerably smaller range (less than one order of magnitude within study type, though the range across field and lab studies is still about 2 orders of magnitude; pink and brown bars for Types A and B in Figure 3). However, as noted earlier, there are fewer biosolids-related data points, which may account for the lesser variation.

Compared to soils, sediments span a much larger range in reported values, primarily due to the field studies (log K_d approximately -0.7 to 4.9). Median log K_d values reported for sludge and SPM are generally higher, though of similar variability. However, the number of studies reporting log K_d in sludge or SPM are few compared to soils or sediments.

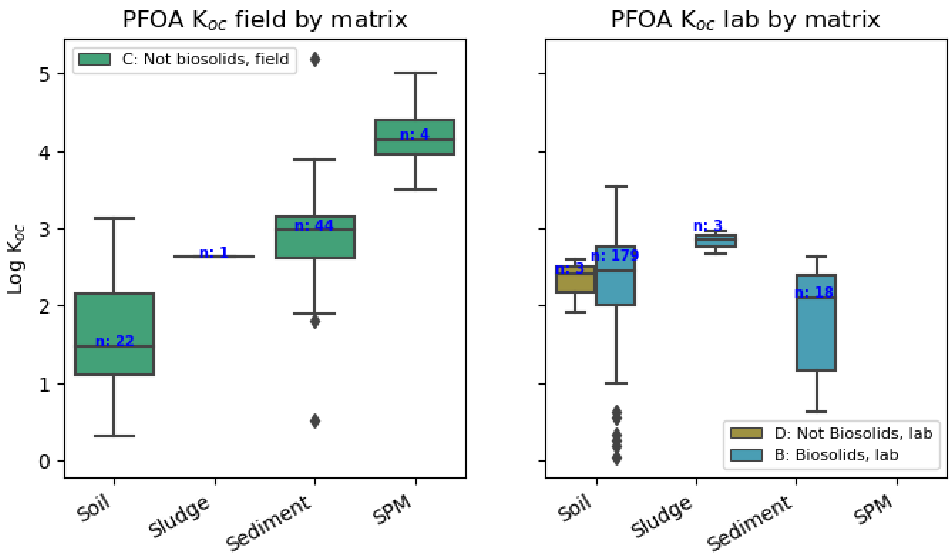


Shaded portion represents the interquartile range (IQR); whisker ends are quartile ±1.5xIQR; diamonds are outliers.
SPM = suspended particulate matter.

Figure C-20. Boxplots of log K_d values for PFOA by matrix for field studies (left) and lab studies (right).

As shown in **Figure C-21**, overall log K_{oc} values in soil and sediment from field and lab studies are similarly variable as log K_d values, covering about three orders of magnitude for soil (log K_{oc} approximately 0 to 3.5) and nearly five orders of magnitude in sediment (log K_{oc} approximately 0.5 to 5.2). No literature values were identified for log K_{oc} measured in field studies of soils. However, laboratory studies of log K_{oc} measurements in soil were reported in both biosolids and non-biosolids studies. The reported results for soils from both biosolids and non-biosolids are comparable (Figure C-21, plot on right) although the range of non-biosolids reported results are much larger (0 to approximately 3.5).

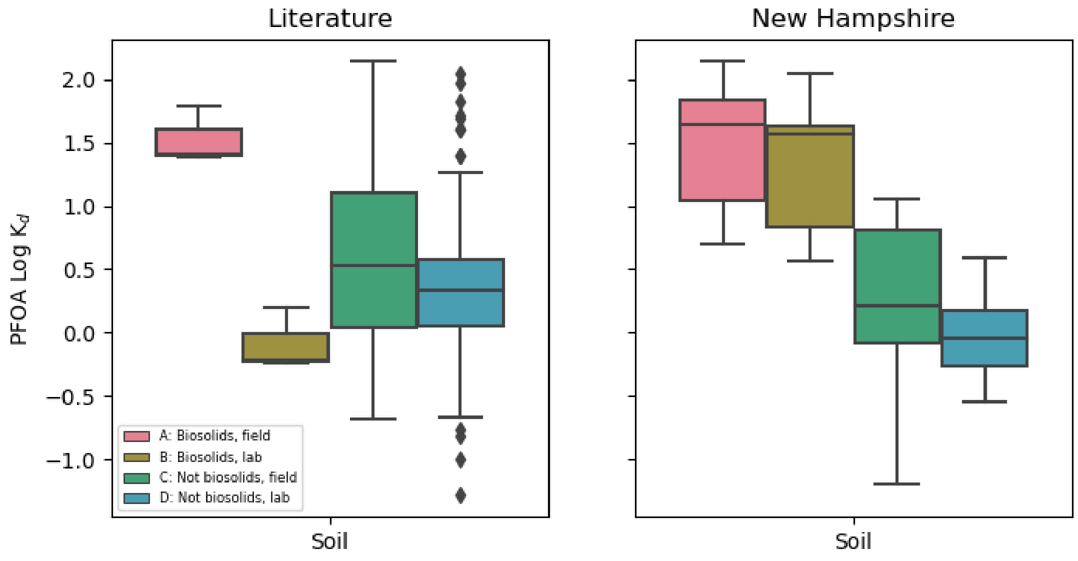
Reported log K_{oc} values for sludge are considerably less variable (log K_{oc} approximately 2.6 to 3). Values for SPM are more variable than sludge and somewhat less variable than soil and sediment. The SPM values are considerably higher than other samples of the same type (*i.e.*, field) but are based on only four studies (all field), so additional data may be needed to assess if there is a difference between log K_{oc} values in SPM compared to other media.



Shaded portion represents the interquartile range (IQR); whisker ends are quartile ±1.5xIQR; diamonds are outliers.
SPM = suspended particulate matter.

Figure C-21. Boxplots of log K_{oc} values for PFOA by matrix for field studies (left) and lab studies (right).

Figure C-22 compares log K_d values reported in literature for soils alongside those reported by USGS for NH (Tokranov et al., 2023). The range of log K_d values reported is broken down by study type (Types A through D) to facilitate closer inspection. Generally, the range of reported log K_d values in literature and by the USGS compare well and span two orders of magnitude (log K_d approximated –0.5 or –1 to approximately 2). Additionally, in both datasets, the median log K_d values for field studies of biosolids are higher than those for non-biosolids or laboratory studies.

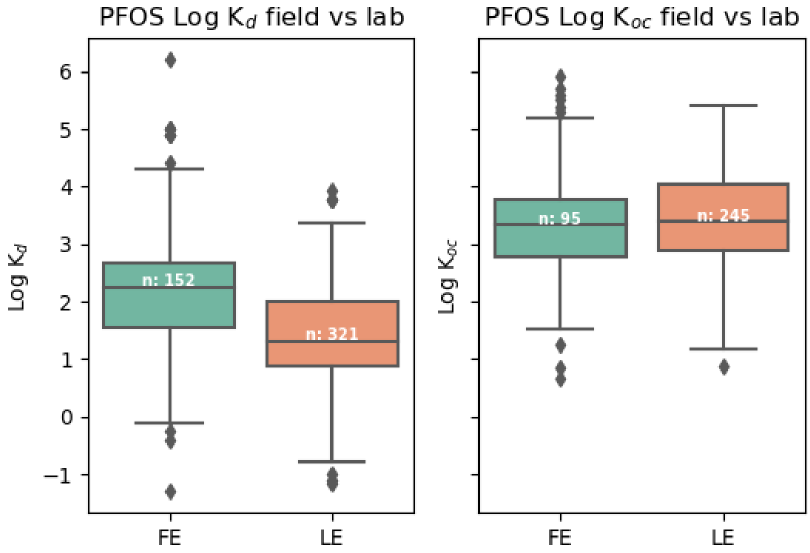


Shaded portion represents the interquartile range (IQR); whisker ends are quartile ±1.5xIQR; diamonds are outliers.

Figure C-22. Boxplots of log K_d values in soil for PFOA by from literature and USGS/NH.

Results for PFOS

As was found for PFOA, reported log K_d and log K_{oc} values for PFOS in literature span approximately six and five orders of magnitude, respectively (**Figure C-23**). For log K_d, the median value from field data and lab data are similar (this is including both biosolids and non biosolids studies). Unlike log K_{oc}, the median log K_d value for field data is somewhat more than the median for laboratory data. The overall range of reported results for log K_{oc} and log K_d are larger for field studies as compared to laboratory studies. This maybe because more observations are derived from laboratory studies than field studies.



Shaded portion represents the interquartile range (IQR); whisker ends are quartile ±1.5xIQR; diamonds are outliers.

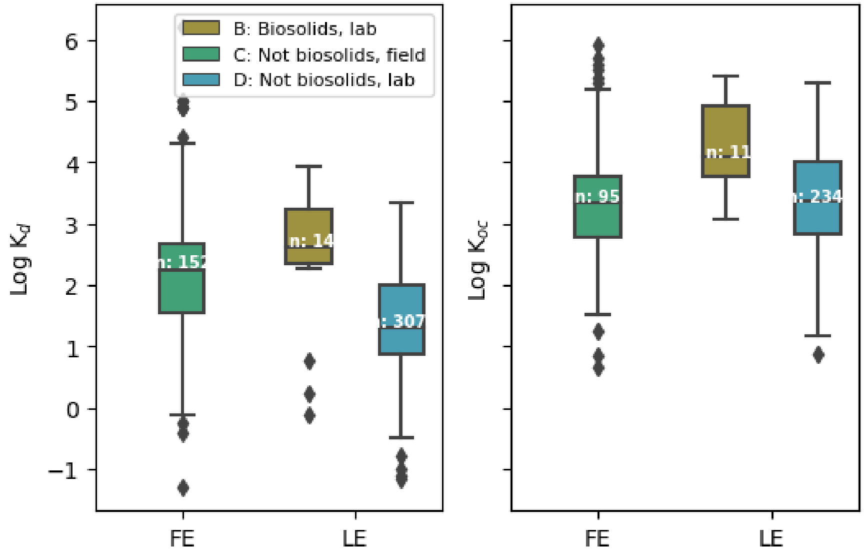
Figure C-23. Boxplots of log K_d (left) and log K_{oc} (right) values for PFOS: field (FE) vs. laboratory (LE) studies.

As shown in **Figure C-24**, the range of log K_d values (left side) based on lab data is similar for studies related to biosolids (Type B; brown box) and studies not related to biosolids (Type D; blue box), despite there being more non-biosolids data. For log K_{oc} (right side, **Figure C-24**), the lab data associated with biosolids (Type B; brown box) appears to be a subset of the range of reported values for non-biosolids studies (Type D; blue box). No field studies of biosolids (Type A) were identified for either log K_d or log K_{oc}. However, the non-biosolids field studies (Type C; green box, **Figure C-24**) spans a large range that encompasses results for other study types (both biosolids and non biosolids lab studies, Types B and D).

We further evaluated the data by sample media: soil, sludge, sediment, or suspended particulate matter (SPM). Most log K_d and log K_{oc} studies have been performed in soil (n=309 for K_d, n=253 for K_{oc}) or sediment (n=133 for K_d, n=78 for K_{oc}), with a lesser number in sludge (n=24 for K_d, n=4 for K_{oc}), and very few in SPM (n=7 for K_d, n=5 for K_{oc}).

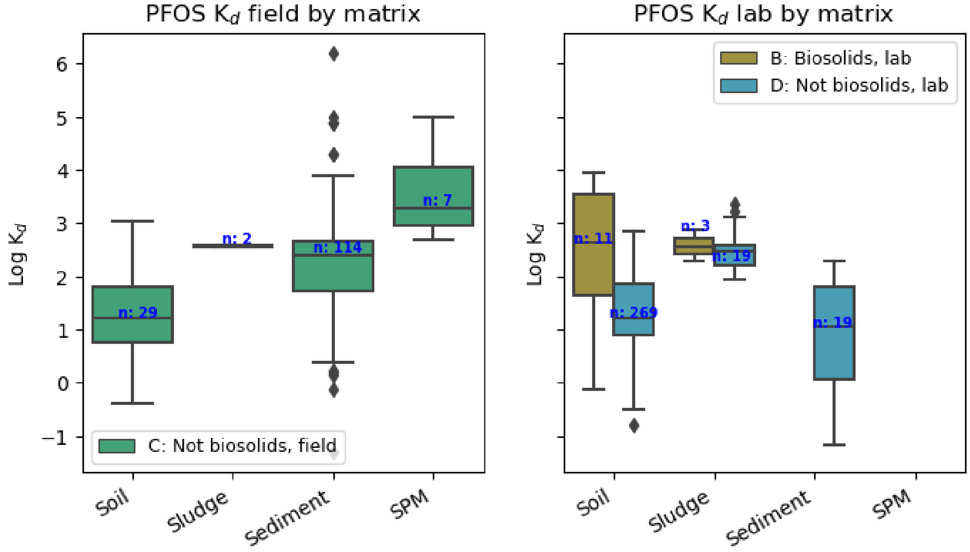
As shown in **Figure C-25**, the overall range of log K_d values in soil (approximately -0.8 to 3.9) is narrower than the range of log K_d values in sediment (approximately –1.3 to 6.2). Note that the soil values are all lab studies, as no field studies (in biosolids or otherwise) were identified for soil.

Compared to soils and sediments, log K_d values reported for sludge and SPM are generally less variable. Reported log K_d values for sludge range from about 2 to 3.4; studies with SPM reported much higher log K_d values (greater than approximately 3 to 5). However, only seven field studies reported log K_d in SPM; additional measurements may be needed to assess if there is a difference between log K_d values in SPM compared to other media.



Shaded portion represents the interquartile range (IQR); whisker ends are quartile ±1.5xIQR; diamonds are outliers. FE=field, LE=lab

Figure C-24. Boxplots of log K_d (left) and log K_{oc} (right) values for PFOS: biosolids vs. not biosolids.

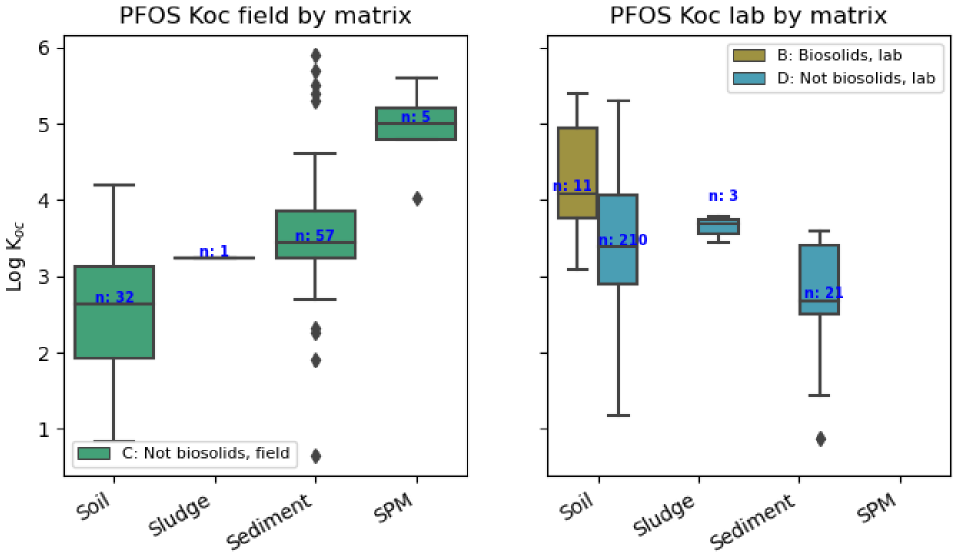


Shaded portion represents the interquartile range (IQR); whisker ends are quartile ±1.5xIQR; diamonds are outliers. SPM = suspended particulate matter.

Figure C-25. Boxplots of log K_d values for PFOS by matrix for field studies (left) and lab studies (right).

As shown in **Figure C-26**, the log K_{oc} values in soil span about four orders of magnitude, from about 1 to 5.4, while sediment log K_{oc} values vary more, about five orders of magnitude, from about 0.7 to 5.9. Closer inspection reveals that the larger range in sediment log K_{oc} values is due to the very wide range (including many outliers) in reported values from field studies that are not biosolids related (green box for sediment, Figure C-26 left panel). Biosolids-related laboratory studies for sediments report a much smaller range of log K_d values approximately between 1 and 3.6 but this maybe an artifact of the number of available studies or perhaps less variability of sediment conditions in the lab compared to the field. There are fewer biosolids-related laboratory study reported values (n=21) in comparison to field studies (n=57).

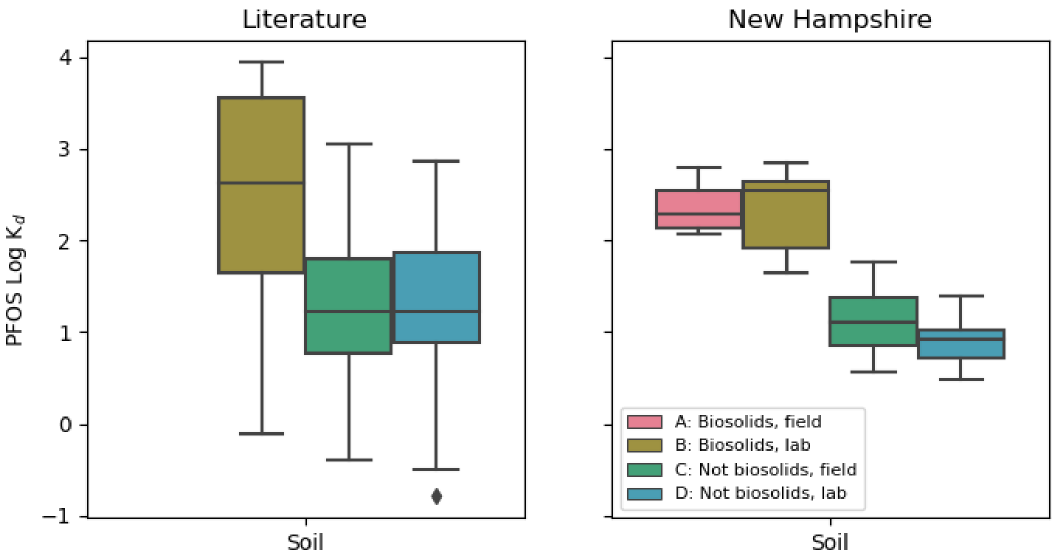
Compared to soils and sediments, log K_{oc} values reported for SPM, where available, are generally higher (Figure C-26). However, the number of studies reporting log K_d in SPM (n=5) are much fewer than soils (n=253) or sediments (n =78).



Shaded portion represents the interquartile range (IQR); whisker ends are quartile ±1.5xIQR; diamonds are outliers. SPM = suspended particulate matter.

Figure C-26. Boxplots of log K_{oc} values for PFOS by matrix for field studies (left) and lab studies (right).

Figure C-27 compares log K_d values reported in literature for soils alongside those reported by USGS for NH (Tokranov et al., 2023). The range of log K_d values reported is broken down by study type (Type A through D), where available, to facilitate closer inspection. Generally, the range of reported log K_d values in literature and by the USGS compare well, but the NH data spans a smaller range (log K_d between 0.5 and approximately 3) in comparison to the literature dataset (log K_d between -0.8 and 4). Additionally, in both datasets, while median log K_d values for biosolids laboratory studies are higher than those for non-biosolids studies, they are similar when comparing for the same conditions and content which was not the case for PFOA.

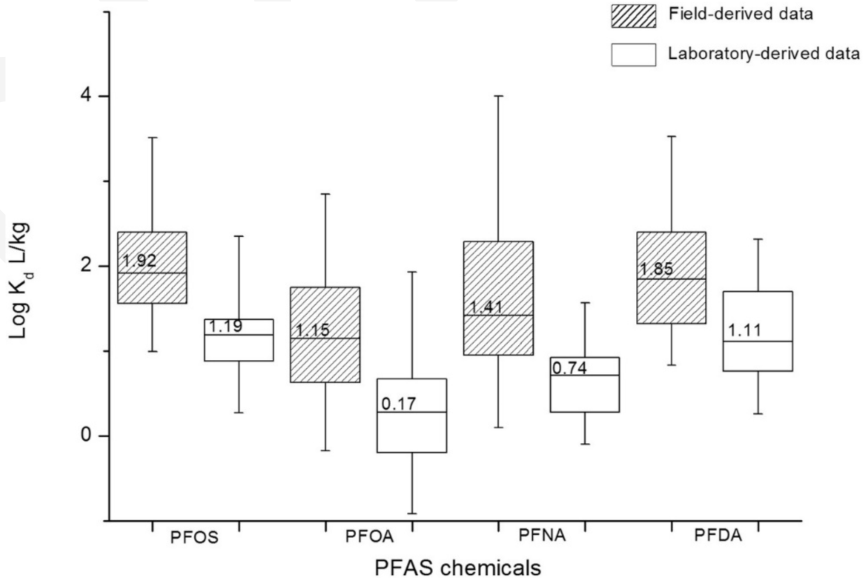


Shaded portion represents the interquartile range (IQR); whisker ends are quartile ±1.5xIQR; diamonds are outliers.

Figure C-27. Boxplots of log K_d values in soil for PFOS by from literature and USGS/NH.

C.3.2.5 Conclusions

Overall, the results for log K_d found in the literature remain consistent with older review papers. For example, Li et al. (2018) presents quartile plots of log K_d for PFOA and PFOS based on data retrieved from 28 peer-reviewed articles and reports spanning 2001 to 2017 (**Figure C-28**). These authors reported that measured values from the field for log K_d are greater than laboratory data, and log K_d values for PFOS are greater than PFOA. Median values from Li et al. (2018) agree very well with data presented in this review. A comparable plot for log K_{oc} was not identified in the literature.



Shaded portion represents the interquartile range (IQR); whisker ends are quartile ±1.5xIQR; diamonds are outliers.

Figure C-28. Data from Li et al. (2018) showing median values of log K_d in field and laboratory studies.

The trend of field studies yielding higher values than laboratory studies was consistent everywhere except for log K_{oc} for PFOS (Figure C-23, right side); several reasons may account for this, including an artifact of the field vs. lab classification scheme used in this analysis, a difference in the number of data points between field (n=95) and lab studies (n=245) or due to some other unidentified reason.

Although the number of biosolids-associated data points are lower than non-biosolids, the range of biosolids-associated values are usually captured within the range of non-biosolids oriented studies. The spread and median values of log K_{oc} tend to show less variability across the various categories of field and laboratory studies, biosolids and non-biosolids related studies, and across media than log K_d. This behavior is reasonable as log K_d additionally reflects the effects of organic carbon (OC) variability in various matrices, as well as other parameters (e.g., pH), whereas log K_{oc} does not.

In terms of using the information gathered in this review for predictive purposes within the current risk assessment framework, log K_{oc} would be preferred for several reasons. First, sampling a matrix-specific (soil, sediment, or SPM) value of K_d with an implicit organic content value would not likely be the same as the organic carbon content in the same simulated matrix based on soil survey information (e.g., gSSURGO[18]). Media matching of K_d would also be limited to soils and sediments. Using K_{oc} would remove that potential inconsistency, letting the matrix organic carbon content determine the value of K_d, and there is more data available on organic content on a national scale than media specific values of K_d.

Consequently, K_{oc} was used as an input parameter to EPACMTP along with estimated location-specific fraction organic carbon (discussed below) to compute location-specific solid phase adsorption coefficient, K_d. All simulations were performed using a “low-Koc” and a “high-Koc” value for both PFOA and PFOS (**Table C-9**). The “low-Koc” and “high-Koc” values represent the closest-reported literature values corresponding to the 10th percentile and the 90th percentile values of the corresponding distribution for the soil matrix.

Table C-9. EPACMTP input parameter values for PFOA and PFOS organic carbon partition coefficient

Scenario	PFOA Koc (cm^3/g)	PFOS Koc (cm^3/g)
Low-K_{oc}	26 (Hubert, M. et al., 2023)	250 (Johnson et al., 2007)
High-K_{oc}	1,100 (Campos-Pereira, H. et al., 2023)	22,000 (Chen, X. T. et al., 2020)

C.3.3 Environment-specific Parameters

EPACMTP requires information about soil and aquifer properties as model inputs.

C.3.3.1 Soil Properties

For soils, EPACMTP uses soil texture as a key to generate consistent hydrological properties for the unsaturated zone model. The primary data source for soil properties was the Soil Survey Geographic (SSURGO) database. SSURGO is a repository of nationwide soil properties collected by the National Cooperative Soil Survey over the last century (USDA, 2017). SSURGO data were collected at scales ranging from 1:12,000 to 1:63,360 and are linked to map unit polygons ranging between 1 and 10 acres. These map units provide the finest spatial resolution and span most of the conterminous United States. Soil attributes linked to these map unit polygons are stored within a relational database broken out by soil component and soil horizon. Each map unit contains data on the prevalence of each component and horizon within the map unit. **Table C-10** shows the crosswalk used to assign the SSURGO detailed soil textures to basic Soil Conservation Service (SCS) textures, and then to the EPACMTP mega textures. SSURGO soils are classified into 21 texture classes, which map to 12 SCS textures. EPACMTP uses three

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[18] https://www.nrcs.usda.gov/resources/data-and-reports/gridded-soil-survey-geographic-gssurgo-database

soil mega textures to represent the variability of hydrologic soil properties, so each SSURGO soil texture was cross walked to the EPACMTP mega texture with the most similar hydrogeologic properties.

The dominant soil texture was estimated by computing the percentages of the three mega-textures (Silty Clay Loam, Silty Loam and Sandy Loam) within a 5-mile radius of each geographic location: Boulder (lat/long: 40.037361, -105.228139), Chicago (lat/long: 41.979444, -87.904444) and Charleston (lat/long: 32.898611,-80.040833). As shown in the Unsaturated Zone section of **Appendix B**, site-specific soil texture model inputs reflect the dominant mega texture of Sandy Loam at Boulder, Silty Clay Loam for Chicago and Silty Loam for Charleston. These model inputs include saturated hydraulic conductivity (SATK), van Genuchten soil moisture parameters (ALPHA and BETA), residual and saturated water contents (WCR and WCS, respectively), percent organic matter (POM), and soil bulk density.

Table C-10. Soil Texture Crosswalk

Detailed SSURGO Soil Texture	Basic SCS Texture	EPACMTP Soil Mega texture
Loamy Sand	Loamy Sand	Sandy Loam
Loamy Coarse Sand		
Loamy Fine Sand		
Loamy Very Fine Sand		
Sand	Sand	
Coarse Sand		
Fine Sand		
Very Fine Sand		
Sandy Loam	Sandy Loam	
Coarse Sandy Loam		
Fine Sandy Loam		
Very Fine Sandy Loam		
Silt Loam	Silt Loam	Silt Loam
Silt	Silt	
Loam	Loam	
Sandy Clay Loam	Sandy Clay Loam	
Clay Loam	Clay Loam	
Silty Clay Loam	Silty Clay Loam	Silty Clay Loam
Sandy Clay	Sandy Clay	
Silty Clay	Silty Clay	
Clay	Clay	

C.3.3.2 Hydrogeologic Environment

Each location modeled in this analysis was assigned a hydrogeologic environment from EPA's Hydrogeologic Database (HGDB) to characterize four subsurface parameters required by EPACMTP: depth to ground water, aquifer thickness, hydraulic gradient, and saturated hydraulic conductivity (see Appendix B). The HGDB was developed by the American Petroleum Institute (Newell et al., 1989; 1990) to specify correlated empirical probability distributions of these four parameters for the 12 distinct hydrogeologic environments described in Newell et al. (1990).

To assign appropriate aquifer conditions to each unit's geographic location, EPA first developed a national geographic map of the 12 hydrogeologic environments (**Figure C-29**). The following individual map layers were combined using GIS software to develop a single map layer for assigning the 12 hydrogeologic environments across the United States:

Shallowest principal aquifers from Principal Aquifers of the Conterminous United States, Hawaii, Puerto Rico, and the US Virgin Islands [USGS map file: aquifrp025]. 1:2,500,000 map scale, was used as the base layer in the assessment and to delineate several of the 12 hydrogeologic environments.

Alluvial and glacial aquifers from Aquifers of Alluvial and Glacial Origin [USGS map file: alvaqfp025]. 1:2,500,000 map scale, was used to represent alluvial and glacial aquifers for the 22 states north of the southernmost line of glaciation. Note that the alluvial aquifers in this coverage are identical to those in the Hunt (1979) surficial geology layer below.

Surficial geology of the conterminous United States was taken from:

- Surficial Geology of the Conterminous United States [map file: geol75m]. 1:7,500,000 map scale, provided by Hunt (1979), these data were used to characterize shallow soil lithology and alluvial aquifers.
- The Surficial Deposits and Materials in the Eastern and Central United States (East of 102 degrees West Longitude) [map file: sfgeoep020]. 1:1,000,00 map scale, includes the line of maximum glacial advance and represents surficial materials that accumulated or formed during the past two million years, including residual soils, alluvium, and glacial deposits.

Karst aquifers from Engineering Aspects of Karst [map file: karst0p075], 1:7,500,000-map scale, showing karst and pseudokarst (*i.e.*, karst-like terrain produced by processes other than the dissolution of rocks) across the United States.

Bedrock geology from Generalized Geologic Map of the United States [map file: geolgyp075], 1:7,500,00 map scale, showing the bedrock geology at or near land surface (*i.e.*, beneath surficial soils, alluvium and glacial deposits).

STATSGO soils, 1:250,000 map scale, from the digital map and attribute data for soils.

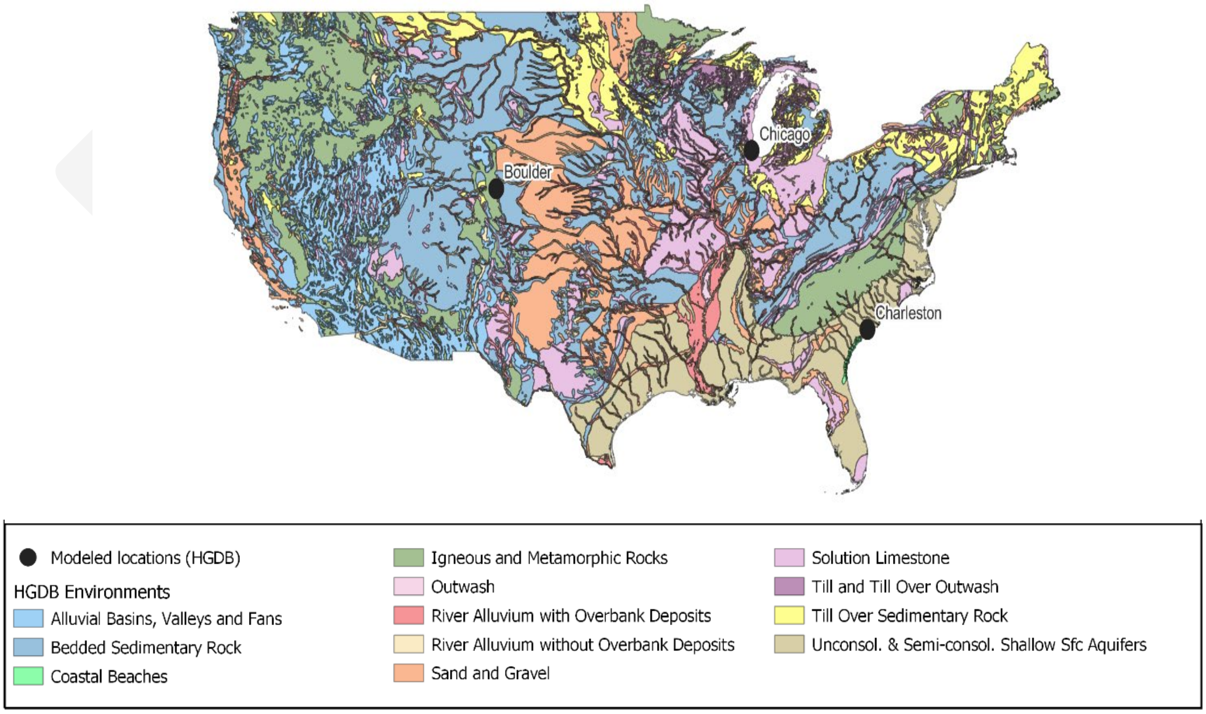

Figure C-29. National geographic map of the 12 hydrogeologic environments developed by EPA.

To create the hydrogeologic environment layer, each individual data layer described above was obtained as a GIS shapefile and processed, as needed, to ensure that coordinate systems matched and the layers could be overlain. Additional details of the data used to parameterize the unsaturated zone and the development and use of the HGDB are given in the *EPACMTP Parameters/Data Background Document* (US EPA, 2003b). The national hydrogeologic environment layer developed in GIS was used for assigning an aquifer type to each geographic location of interest: Boulder (lat/long: 40.037361, -105.228139), Chicago (lat/long: 41.979444, -87.904444) and Charleston (lat/long: 32.898611,-80.040833). Given an aquifer code setting for each application unit, a correlated sample of key aquifer model input parameters (hydraulic conductivity, hydraulic gradient, depth to the water table, and saturated thickness) was selected from a population of samples taken from similar hydrogeological settings.

C.3.3.3 Other Calculated Environment-Specific Input Parameters

Unsaturated Zone Longitudinal Dispersivity

Dispersion is the phenomenon by which a dissolved constituent in soil or ground water is mixed with uncontaminated water and becomes reduced in concentration at the perimeter of the plume. Not all of a constituent is traveling at the same velocity, due to differences in pore size and flow path length and friction along pore walls, resulting in mixing along the flow path which decreases solute concentrations. Note that the unsaturated zone longitudinal dispersivity is measured along the path of flow in the downward direction. For the current risk assessment, longitudinal dispersivity is calculated as a linear function of the total depth of the unsaturated zone according to the following equation which is based on a regression analysis of data presented by EPRI (1985) and has a correlation coefficient of 0.66:

$$\alpha_{Lu} = 0.02 + 0.022\ D_u$$

where,

α_{Lu} = longitudinal dispersivity (m)
D_u = total depth of the unsaturated zone (m).

Saturated Zone Longitudinal, Transverse and Vertical Dispersivity

The longitudinal dispersivity is the characteristic length that defines spatial extent of dispersion of contaminants, measured in the longitudinal direction, that is, along the flow path or in the X-direction. The longitudinal dispersivity is calculated using equation 5.11 of the *EPACMTP Parameters/Data Background Document* (US EPA, 2003b) based on a receptor well distance of 30 meters and a reference dispersivity corresponding to 1 meter. The horizontal transverse dispersivity is calculated as 1/8th the longitudinal dispersivity in accordance with equation 5.13 of the *EPACMTP Parameters/Data Background Document* (US EPA, 2003b). Similarly, the vertical dispersivity is calculated as 1/160th the longitudinal dispersivity in accordance with equation 5.14 of the *EPACMTP Parameters/Data Background Document* (US EPA, 2003b).

Fraction organic carbon

The fraction organic carbon in the soil or aquifer is estimated from the location-specific percentage of organic matter (see Table B-9, Appendix B) by dividing it by a factor of 174 in accordance with equation 3.3 of the *EPACMTP Parameters/Data Background Document* (US EPA, 2003b).

Recharge

Recharge is water percolating through the soil to the aquifer outside the footprint of the unit. Typically, EPACMTP selects a recharge rate using a meteorological station assignment (based on the geographic location and topography of a unit setting) and by the unit's associated soil texture mentioned above. Using the soil texture and station assignment, a recharge rate was computed using the HELP model (US EPA, 2020) and using the nearest OPP synthetic weather data (Fry et al., 2016). Further details about how these rates were determined and other options for determining recharge rates outside of the EPACMTP model can be found in the *EPACMTP Parameters/Data Background Document* (US EPA, 2003b).

C.4 Model Outputs

The output of EPACMTP is a prediction of the contaminant concentration arriving at a downgradient groundwater receptor location, and is a time-dependent concentration, corresponding to the finite source scenario. The model can calculate both the peak concentration arriving at the well and maximum time-averaged concentrations. In this analysis, peak concentrations were used to develop human risk estimates. Because the subsurface migration of PFOA and PFOS may be very slow, it may take a long time for the plume to reach the receptor well, and the maximum exposure may not occur until a very long time after the land application ceases. For example, the peak arrival time for PFOA and PFOS at the receptor well varies between 34 years and 9974 years across the modeled locations, scenarios and chemicals. Therefore, for this analysis, maximum exposures occurred within EPACMTP's maximum default time horizon of 10,000 years.

Table C-11 provides a summary of the simulated PFOA and PFOS exposure concentrations at a receptor well located at 5 meters from the edge of the farm field (*i.e.*, center of the buffer). As noted in **Section C.3**, the exposure concentration for this evaluation is the maximum of the peak concentrations at the receptor well across four well depths (0.5 m, 1.0 m, 1.5 m and 2.0 m below the water table). Exposure concentrations reported in Table C-11 were used as inputs to calculate risks for the groundwater pathway.

Table C-11. Groundwater Pathway Receptor Exposure Concentrations: Maximum of the Peak Concentrations at a Receptor Well Located at the Center of the Buffer across Four Well Depths (mg/L)

Scenario	Low Koc			High Koc		
	Dry	Average	Wet	Dry	Average	Wet
PFOA						
LAU (Crop)	1.7E-03	8.8E-03	7.4E-03	4.4E-34	4.1E-15	1.2E-06
LAU (Pasture)	4.1E-03	5.8E-03	3.2E-03	5.2E-34	2.9E-06	1.5E-05
LAU (Reclamation)	4.0E-03	1.2E-03	4.0E-03	1.8E-35	3.9E-11	5.9E-09
SDU (No Liner)	1.9E-02	1.3E-02	1.3E-02	4.7E-11	7.6E-15	3.4E-07
SDU (Clay Liner)	1.7E-02	8.3E-03	9.8E-03	8.7E-14	8.0E-16	1.7E-07
SDU (Composite Liner)	1.7E-05	1.6E-06	3.8E-06	2.8E-35	3.0E-35	4.5E-35
PFOS						
LAU (Crop)	1.2E-14	1.8E-05	2.3E-05	8.3E-35	7.8E-35	5.7E-16
LAU (Pasture)	1.6E-16	8.2E-05	4.7E-04	1.4E-34	1.2E-15	1.3E-07
LAU (Reclamation)	1.5E-06	2.2E-06	6.5E-06	5.2E-36	1.6E-36	1.7E-08
SDU (No Liner)	1.5E-05	4.4E-06	4.4E-05	1.9E-36	1.9E-36	8.0E-13
SDU (Clay Liner)	1.1E-05	2.9E-06	2.0E-05	1.9E-36	1.9E-36	1.0E-15
SDU (Composite Liner)	6.9E-34	6.8E-34	6.6E-34	1.9E-36	2.0E-36	2.9E-36

C.5 References

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APPENDIX D. SENSITIVITY ANALYSIS

D.1 Introduction

Sensitivity analysis is the evaluation of model input parameters to see how they affect model outputs, thereby providing a fundamental understanding of the simulated system (Reilly and Harbaugh, 2004). In the current study, a sensitivity analysis was performed to understand the sensitivity of downstream models and predicted risk outputs to individual constituent-specific and environment-specific parameters. The sensitivity of predicted risk outputs from inputs to two downstream models, EPACMTP and VVWM are discussed here.

D.2 Methods

The sensitivity analysis of the EPACMTP model was conducted such that the model input value for a single parameter was varied at a time and the change in the ratio of initial PFOA and PFOS source concentrations in biosolids leachate to predicted PFOA and PFOS concentrations at an observation well located in the center of the buffer, at 5 meters from the source. The ratio of initial PFOA and PFOS source concentrations in biosolids leachate to predicted PFOA and PFOS concentrations at an observation well is directly proportional to the predicted risk outputs and is therefore an appropriate metric to understand model sensitivity.

In contrast, the sensitivity analysis for VVWM was targeted by reviewing the governing fate and transport equations within the model, ignoring irrelevant pathways for PFOA and PFOS (e.g., degradation) and only testing sensitivity of parameters from appropriate pathways.

Additional discussion on reducing relevant pathways and parameter selection is provided below.

D.2.1 EPACMTP

Model sensitivity was tested for environment- and chemical-specific input parameters for both PFOA and PFOS using three source models or scenarios including crop, pasture and surface impoundment at multiple geographic locations as described below.

D.2.1.1 Unsaturated and Saturated Zone Parameters

For each chemical and source model, model sensitivity was tested at two locations, Boulder and Charleston, with bounding meteorological conditions of dry and wet, respectively and assuming three representative source models, crop, pasture and surface impoundment/disposal unit with clay liner. For each chemical, source model and location, model sensitivity was tested one-at-a-time for 13 EPACMTP input parameters consisting of 8 unsaturated zone and 5 saturated zone parameters. A total of 312 EPACMTP model simulations were performed for this sensitivity analysis.

The 13 EPACMTP input parameter sensitivities tested are listed in **Table D-1**. For each parameter, bounding values were selected from the cumulative frequency distribution generated by performing a representative nationwide landfill modeling analysis using the regional site-based modeling methodology as reported in the EPACMTP *Parameters/Data Background Document* (US EPA, 2003). Selection of bounding values from this cumulative frequency distribution for input parameter sensitivities is appropriate as it reflects nationwide variability of these parameters. For each parameter, the lower and upper bound values were selected as the 10th and 90th percentile of the cumulative frequency distribution, respectively. However, adjustment of some parameter inputs from the 10th percentile lower bound value were made to the crop and pasture source model scenarios because these parameter combinations resulted in the violation of underlying assumptions of the model (e.g., excessive water table mounding). Adjusted values for these parameter inputs are also noted in **Table D-1**.

Table D-1. Saturated and Unsaturated Zone Parameters Tested and Corresponding Lower and Upper Bound Values Tested

Parameter	EPACMTP Model Code	Units	Lower bound (10th %ile)	Upper bound (90th %ile)	Reference[a]
Unsaturated Zone					
Saturated hydraulic conductivity	US01	cm/hr	6.79E-03	1.93E+00	Table 5.5
Van Genuchten alpha parameter, a soil-specific shape parameter	US02	cm^{-1}	5.96E-03	5.90E-02	Table 5.7
Van Genuchten beta parameter, a soil-specific shape parameter	US03	unitless	1.20E+00	1.82E+00	Table 5.8
Residual water content	US04	unitless	4.89E-02	9.37E-02	Table 5.9
Saturated water content (effective porosity)	US05	unitless	4.10E-01	4.50E-01	Table 5.10
Depth from ground surface to water table	US06	m	1.68E+00 3.96E+00[b]	4.27E+01	Table 5.2
Percent organic matter	US08	%	1.05E-01	2.15E-01	Table 5.12
Bulk density of unsaturated soil	US09	g/cm^3	1.60E+00	1.67E+00	Table 5.11
Saturated Zone					
Effective porosity of aquifer	AS02	unitless	3.00E-05	6.94E-01	Wolff (1982)
Aquifer soil bulk density	AS03	g/cm^3	1.30E+00	1.70E+00	Table 5.18
Thickness of saturated zone	AS04	m	4.27E+00 1.43E+01[c]	9.14E+01	Table 5.20
Hydraulic conductivity of saturated zone (aquifer)	AS05	m/yr	1.73E+02	3.15E+04	Table 5.21
Regional hydraulic gradient in the aquifer	AS07	m/m	9.00E-04	3.10E-02	Table 5.22

a Table references correspond to the EPACMTP Parameters/Data Background Document (US EPA, 2003).
b A lower bound value at the 25th percentile for depth to water table from ground surface was used for simulations in Charleston to account for the simulated water table rising above ground surface resulting in a mounding violation within the model.
c A lower bound value at 50th percentile for saturated zone thickness was used for simulations in Charleston to account for the simulated water table rising above ground surface resulting in a mounding violation within the model

D.2.1.2 Chemical-specific Parameters

Model sensitivity to the PFOA and PFOS organic carbon partition coefficient parameter (K_{oc}, cm^3/g), which is the ratio of a constituent's concentration in a theoretical soil containing only organic carbon to its concentration in the ground water was tested at three locations, Boulder, Chicago and Charleston, representing dry, moderate and wet meteorological condition, respectively and assuming two representative source models, crop and pasture. For each chemical, source model and location, model sensitivity was tested by varying the organic partition coefficient (K_{oc}) one-at-a-time between a “low-K_{oc}”, “representative-K_{oc}” and “high-K_{oc}” value as shown in **Table D-2**.

Table D-2. Values of Organic Carbon Partition Coefficient (koc) Tested for PFOA and PFOS

Sensitivity Scenario	PFOA K_{oc} (cm^3/g)	PFOS K_{oc} (cm^3/g)
Representative-K_{oc}	114.8	371.5
Low-K_{oc}	2.88	2207
High-K_{oc}	19,953	108,081

PFOA and PFOS K_{oc} values of 114.8 cm^3/g and 371.5 cm^3/g, respectively, were used as the “representative” value as these Koc values were reported by EPA in the Health Effects Support Document for PFOA (US EPA, 2016a) and PFOS (US EPA, 2016b). A review of measured Koc and solid phase adsorption coefficient (K_d) values reported in literature for PFOA and PFOS was conducted with and without associations to biosolids or land application of biosolids, for surface and subsurface soils, aquifer materials, and settled and suspended surface water sediments (Appendix C). Based on the results of this review, a “low-K_{oc}” and a “high-K_{oc}” bounding value was developed, representing the upper and lower extremes (*i.e.*, upper and lower whiskers of a box plot). Here, the lower extreme is mathematically represented by the 1st Quartile (25th percentile; Q1) minus 1.5 times the inter-quartile range (IQR = Q3 minus Q1) and the upper extreme is represented by the 3rd Quartile (75th percentile; Q3) plus the IQR. Although the "low-Koc" was intended to be a bounding value, note that in the case of PFOS, the "low-K_{oc}" value of 2206.73 cm^3/g is greater than the "representative-K_{oc}" value of 371.5 cm^3/g. This maybe because of the large range in PFOS Koc reported in literature with an underlying skewed distribution and several outliers identified in the lower end of the distribution (see e.g., Appendix C, Figure C-26).

A total of 36 EPACMTP model simulations were performed for this sensitivity analysis.

D.2.2 VVWM

Model sensitivity was tested for environment- and chemical-specific input parameters for both PFOA and PFOS using two source scenarios (crop and pasture) and two geographic locations to capture the range of meteorological conditions (dry and wet) that govern modeled overland flow rates.

The sensitivity analysis for VVWM was targeted by reviewing the governing fate and transport equations presented in the model documentation (US EPA, 2019) and ignoring those parameters associated with irrelevant pathways for PFOA and PFOS (e.g., volatilization, degradation). The governing equations contain four effective parameters (Equations (5) through (8)) that influence concentrations in the water column and benthic region of the water body: the effective degradation rates of chemical mass in the water column and the benthic region, the mass transfer coefficient describing mass transfer between the water column and benthic region, and ratio of solute holding capacities of the two domains. As PFOA and PFOS are known for being very stable in the natural environment, it was possible to ignore all first order degradation rates in the formulation. Additionally, the dimensions of the index reservoir[19] used in the risk assessment as the receiving water body are not subject to change and, therefore were not examined as sensitive parameters. The flow of water through the reservoir was indirectly examined by evaluating parameter sensitivity for dry and wet meteorological conditions.

Eliminating first order degradation rates from VVWM Equations (5) through (8) yield the following effective parameters:

Hydrologic Washout (Γ_1)

$$\boldsymbol{\Gamma_1 = \frac{Q}{v_1}}$$

where

Q = volumetric flow rate of water out of the littoral compartment[m^3/s]
v_1 = volume of water in littoral compartment [m^3].

As mentioned earlier, the volume of the reservoir will be constant, and flow into and out of the reservoir will be examined indirectly by comparing simulation results for wet and dry meteorological conditions. Therefore, there are no input parameters here to vary directly.

Effective Benthic Region Dissipation (Γ_2)

$$\boldsymbol{\Gamma_2 = \frac{BK_{sed\_2}}{m_{sed\_2}K_{sed\_2} + m_{DOC\_2}K_{DOC\_2} + v_2}}$$

where

---

[19] The index reservoir is based on the standard waterbody parameters for Variable Volume Water Model (VVWM), the waterbody model used to estimate concentrations in surface water (US EPA, 2019; 2020); see Section A.2.3.2.

B = burial rate of sediment [kg/s]
m_{sed_2} = mass of sediment in benthic region [kg]
K_{sed_2} = linear partitioning coefficient for benthic sediments [ml/g]
m_{DOC_2} = mass of dissolved organic carbon (DOC) in benthic compartment [kg]
K_{DOC_2} = linear partitioning coefficient for DOC in benthic region [ml/g]
v_2 = volume of water in benthic compartment [m^3]

The mass of sediments and DOC in the benthic compartment are based on the concentration of each in the volume of that compartment. Therefore, as the volume of the compartment is fixed, the sensitivity of the concentration of sediments and DOC in the benthic compartment will be examined. Partitioning coefficients for sediments and DOC are calculated using K_{oc} and fraction of organic carbon on those sediments and DOC. Sensitivity to K_{oc} will be expressed separately by differences in low K_{oc} values for PFOA and PFOS.

Mass Transfer Coefficient (Ω)

$$\boldsymbol{\Omega = \frac{D}{m_{sed\_2}K_{sed\_2} + m_{DOC\_2}K_{DOC\_2} + v_2}}$$

where

D = water column to benthic dispersion coefficient [m^2/s]

The numerical formulation incorporates the dispersion coefficient into a mass transfer coefficient that relates the overall dispersion through a boundary layer between the littoral and benthic compartments having thickness Δx. This is expressed in the VVWM input parameter D_over_dx, and will therefore be examined for sensitivity.

$$\Theta = \frac{m_{sed\_2}K_{sed\_2} + m_{DOC\_2}K_{DOC\_2} + v_2}{m_{sed\_1}K_{sed\_1} + m_{DOC\_1}K_{DOC\_1} + v_1}$$

where

m_{sed_1} = mass of sediment in water column [kg]
K_{sed_1} = linear partitioning coefficient in suspended sediments in water column [ml/g]
m_{DOC_1} = mass of DOC in water column [kg]
K_{DOC_1} = linear partitioning coefficient for DOC in water column [ml/g].

Sensitivities of sediment and DOC mass and partitioning will be examined through varying the concentration of each and the fraction of organic content on those components.

The VVWM input parameter sensitivities to be tested are listed in **Table D-3**. For each parameter, bounding values were established by increasing and decreasing VVWM default values by an order of magnitude, or nearly so in most cases. Koc bounding values for PFOA and PFOS are adopted from **Table D-2**. Bounding values for the flow rate through the water body are dictated by overland runoff generated by the hydrology module of the land application unit module using meteorologic data for the dry and wet environments.

Table D-3. VVWM Parameters Tested for Sensitivity

Domain	Model Code	Parameter	Units	Scenario	Lower bound	Default[a]	Upper bound
Media	BNMAS	Areal Biomass in Benthic Compartment	g/m^2	NA	0.0006	0.006	0.06
	DOC2	DOC in Benthic Region	mg/L	NA	1	5	20
	DOC1	DOC in Water Column	mg/L	NA	1	5	20
	FROC2	Fraction OC on Benthic Sediments	fraction	NA	0.004	0.04	0.4
	FROC1	Fraction OC on Suspended Sediments in Water Column	fraction	NA	0.004	0.04	0.4
	D_over_dx	Mass Transfer Coefficient	m/s	NA	1E-10	6E-9	1E-8
	PLMAS	Suspended Biomass Concentration	mg/L	NA	0.04	0.4	4.0
	SUSED	Suspended Sediment in Water Column (TSS)	mg/L	NA	10	30	100
Meteorology	Q	Volumetric flow rate[b]	m^3/d	Crop	41	NA	354
				Pasture	17	NA	166
Chemical	Koc	Organic carbon partition coefficient[c]	cm^3/g	PFOA	2.9	114.8	19,953
				PFOS	2,207	371.5	108,081

a These are VVWM defaults for all but Koc; representative Koc values for PFOA and PFOS are described in Appendix C.
b Lower and upper bound are based on long-term annual average values from transient simulations from land application.
c Lower and upper bound Koc values are those presented in Table D-2.

Each bounding parameter value was evaluated for both crop and pasture scenarios under dry and wet conditions for both PFOA and PFOS. A one-at-a-time approach was used to evaluate a bounding value for one parameter for all combinations of biosolids application scenarios (e.g., crop or pasture), and meteorological environments (e.g., dry or wet). All other parameters are represented by VVWM defaults or representative Koc values. Peak concentration values corresponding to the adult receptor, Surface Water pathway, and noncancer benchmarks are used to calculate ratios of concentration corresponding to the bounding value of a parameter to the VVWM default value for the same parameter. To evaluate the sensitivity of flow through the surface water body, ratios of peak concentrations derived from dry and wet meteorology are examined for each chemical and application scenario.

D.3 Results and Discussion

As described earlier, model sensitivity is evaluated by comparing the ratio of initial PFOA and PFOS source concentrations in biosolids leachate to predicted PFOA and PFOS concentrations at an observation well located in the center of the buffer for the lower and upper bound parameter input values for each parameter. The ratio of initial PFOA and PFOS source concentrations in biosolids leachate to predicted PFOA and PFOS concentrations at an observation well is directly proportional to the predicted risk outputs and is therefore an appropriate metric to understand model sensitivity. **Figures D-1 through D-13** present the model sensitivity results for each parameter comparing the three source models or scenarios, chemical (PFOA and PFOS) at two locations, Boulder and Charleston, with bounding meteorological conditions of dry and wet.

D.3.1 EPACMTP

D.3.1.1 Sensitivity to Unsaturated Zone Parameters

The **saturated hydraulic conductivity** of the unsaturated zone soil is a measure of the soil's ability to transmit water under fully saturated conditions. It is used as an input to the unsaturated zone flow module in EPACMTP and is used to calculate the moisture content in the soil under a given rate of leachate infiltration. The difference between the ratio of initial PFOA and PFOS source concentrations in biosolids leachate to predicted PFOA and PFOS concentrations at the observation well at the lower and upper bound values of this input parameter is less than a factor of 1.7 across PFOA and PFOS for all three scenarios and two locations (**Figure D-1**). Therefore, this input parameter is not considered to be sensitive to the predicted model risk outputs.

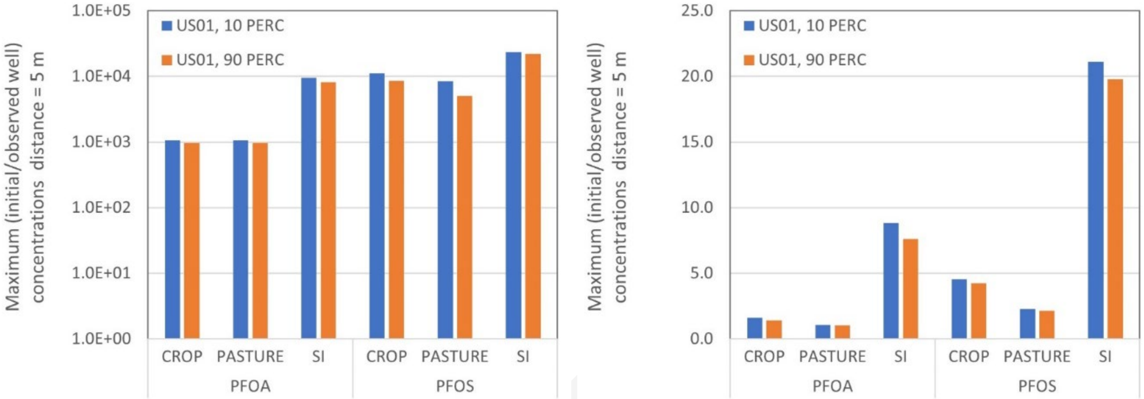


Figure D-1. Sensitivity to saturated hydraulic conductivity of the unsaturated zone for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

The **Van Genuchten shape parameters, alpha and beta**, of the unsaturated zone are soil-specific shape parameters that are obtained from an empirical relationship between pressure head and volumetric water content; are one of the parameters in the van Genuchten (1980) model used for modeling soil-water content as a function of pressure head and are used to calculate the moisture content in the soil under a given rate of leachate infiltration. The van Genuchten parameters alpha and beta are inputs to the unsaturated zone flow module and are used to calculate the moisture content in the soil under a given rate of leachate infiltration. The difference between the ratio of initial PFOA and PFOS source concentrations in biosolids leachate to predicted PFOA and PFOS concentrations at the observation well at the lower and upper bound values of the alpha input parameter is less than a factor of 1.6 across PFOA and PFOS for all three scenarios and two locations (**Figure D-2**). The difference between the ratio of initial PFOA and PFOS source concentrations in biosolids leachate to predicted PFOA and PFOS concentrations at the observation well at the lower and upper bound values of the beta input parameter is less than a factor of 1.3 across PFOA and PFOS for all three scenarios and two locations (**Figure D-3**). Therefore, these input parameters are not considered to be sensitive to the predicted model risk outputs.

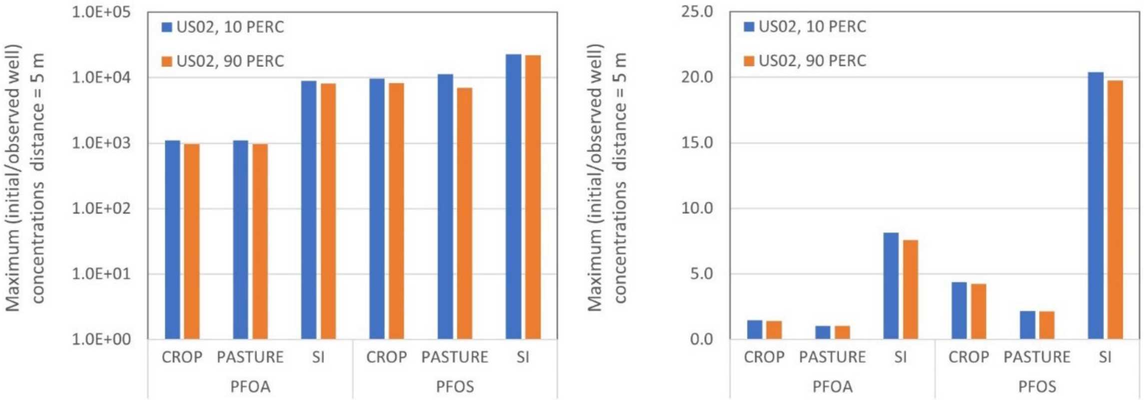


Figure D-2. Sensitivity to van Genuchten parameter alpha for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

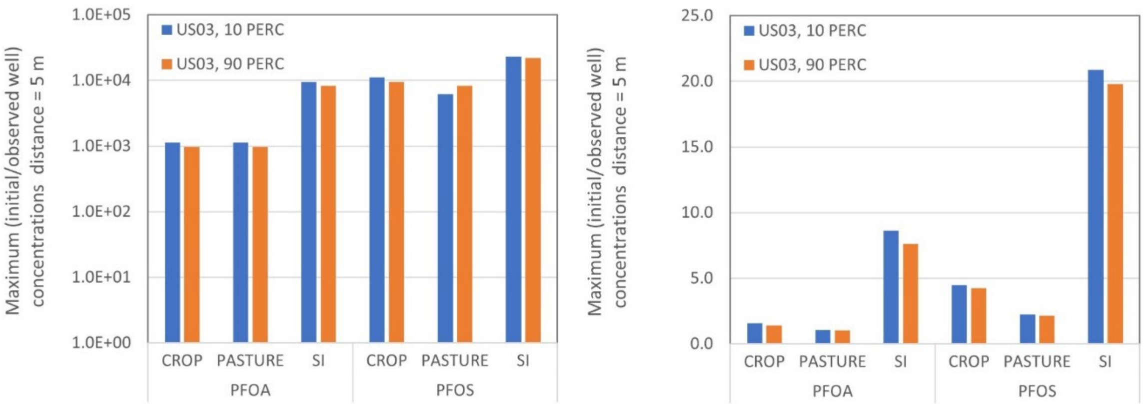


Figure D-3. Sensitivity to van Genuchten parameter beta for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

The **residual water content** is the moisture content of the soil below which a reduction in the pressure head does not result in the loss of moisture. It is an input to the unsaturated zone flow module and is used to calculate the moisture content in the soil under a given rate of leachate infiltration. The difference between the ratio of initial PFOA and PFOS source concentrations in biosolids leachate to predicted PFOA and PFOS concentrations at the observation well at the lower and upper bound values of this input parameter is less than a factor of 1.1 across PFOA and PFOS for all three scenarios and two locations (**Figure D-4**). Therefore, this input parameter is not considered to be sensitive to the predicted model risk outputs.

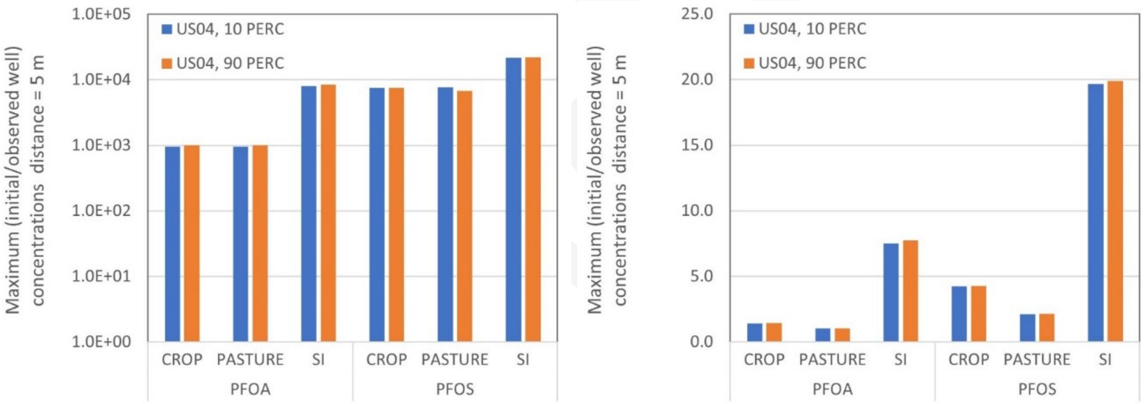


Figure D-4. Sensitivity to residual water content of the unsaturated zone for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

The **saturated water content** represents the maximum fraction of the total volume of soil that is occupied by the water contained in the soil at atmospheric pressure. The difference between the ratio of initial PFOA and PFOS source concentrations in biosolids leachate to predicted PFOA and PFOS concentrations at the observation well at the lower and upper bound values of this input parameter is less than a factor of 1.3 across PFOA and PFOS for all three scenarios and two locations (**Figure D-5**). Therefore, this input parameter is not considered to be sensitive to the predicted model risk outputs.

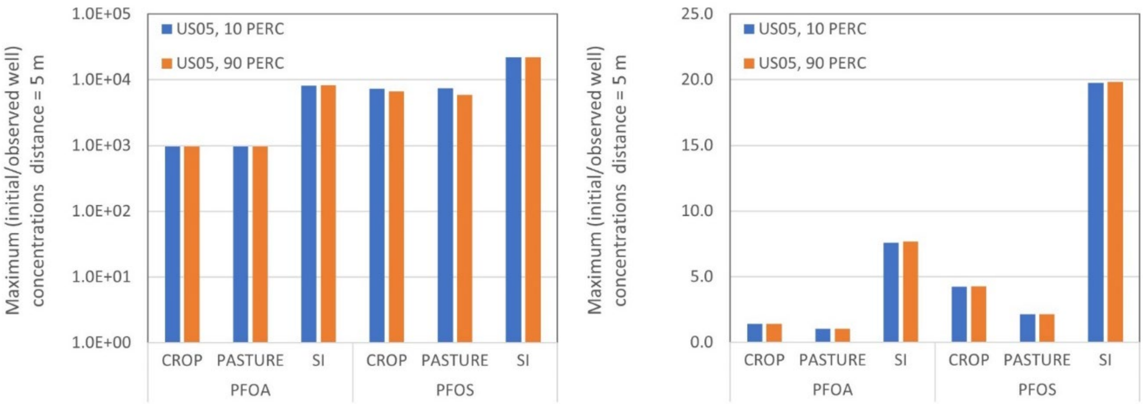
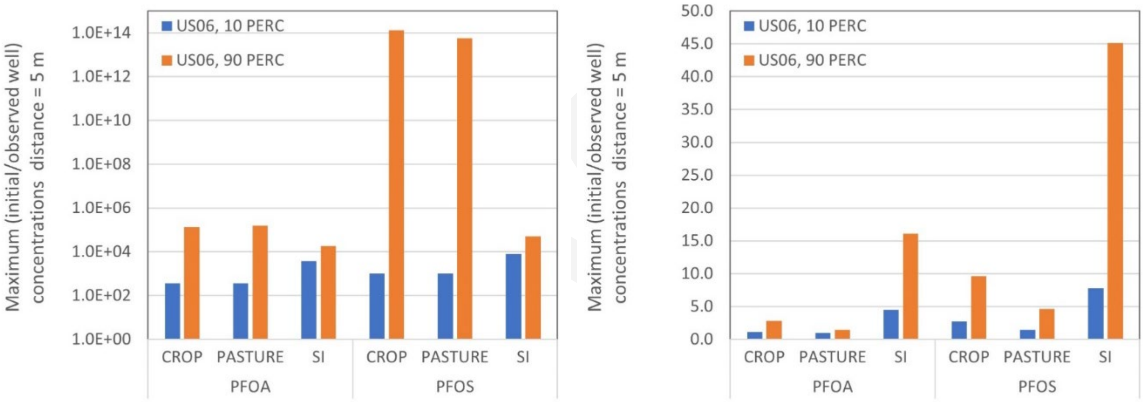


Figure D-5. Sensitivity to saturated water content of the unsaturated zone for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

The **depth from ground surface to water table**, also known as the unsaturated zone thickness, is the vertical distance from the ground surface to the natural water elevation without considering the influence of localized mounding. The EPACMTP model uses the unsaturated zone thickness to determine the travel distance of leachate constituents in the unsaturated zone The difference between the ratio of initial PFOA and PFOS source concentrations in biosolids leachate to predicted PFOA and PFOS concentrations at the observation well at the lower and upper bound values of this input parameter is several orders of magnitude across PFOA and PFOS for all three scenarios and two locations (**Figure D-6**). Therefore, this input parameter is sensitive to the predicted model risk outputs.



Results could not be computed for PFOA/PFOS in Charleston at the 10th percentile depth to water table from ground surface (1.7 m) because the simulated water table was above ground surface resulting in a mounding violation within the model. Therefore, the 25th percentile (3.96 m) was used as the lower bound instead.

Figure D-6. Sensitivity to depth to water table for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

The **percent organic matter** is a measure of the organic material that is present within the soil of the unsaturated zone, measured as a weight percent. EPACMTP converts the percent organic matter in the soil internally to fractional organic carbon content by dividing by a conversion factor of 174. The difference between the ratio of initial PFOA and PFOS source concentrations in biosolids leachate to predicted PFOA and PFOS concentrations at the observation well at the lower and upper bound values of this input parameter is less than a factor of 3.5 across PFOA and PFOS for all three scenarios and two locations (**Figure D-7**). Therefore, this input parameter is not considered to be sensitive to the predicted model risk outputs.

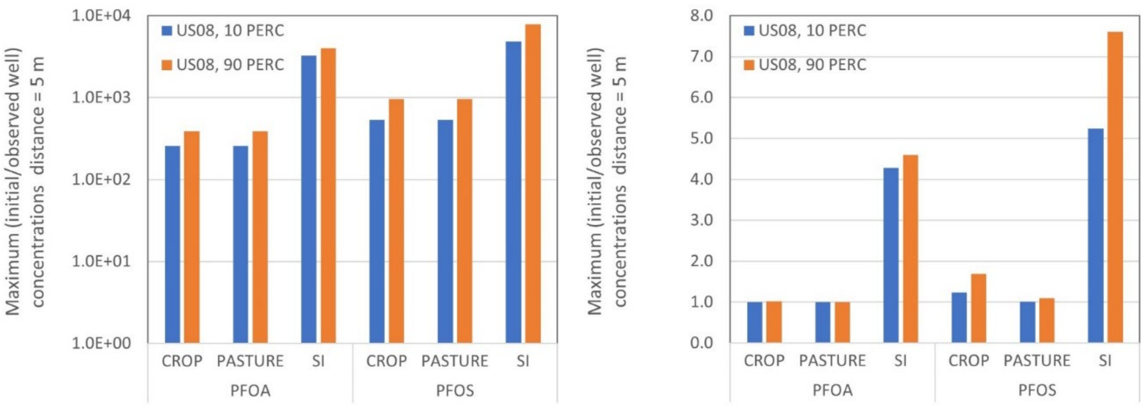


Figure D-7. Sensitivity to percent organic matter of the unsaturated zone for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

The dry **bulk density of the soil** is the ratio of the mass of the solid soil to its total volume. The dry soil bulk density (mass of soil per unit volume) is used to calculate the retardation coefficient of organic constituents and to convert soil mass to volume. The difference between the ratio of initial PFOA and PFOS source concentrations in biosolids leachate to predicted PFOA and PFOS concentrations at the observation well at the lower and upper bound values of this input parameter is less than a factor of 1.2 across PFOA and PFOS for all three scenarios and two locations (**Figure D-8**). Therefore, this input parameter is not considered to be sensitive to the predicted model risk outputs.

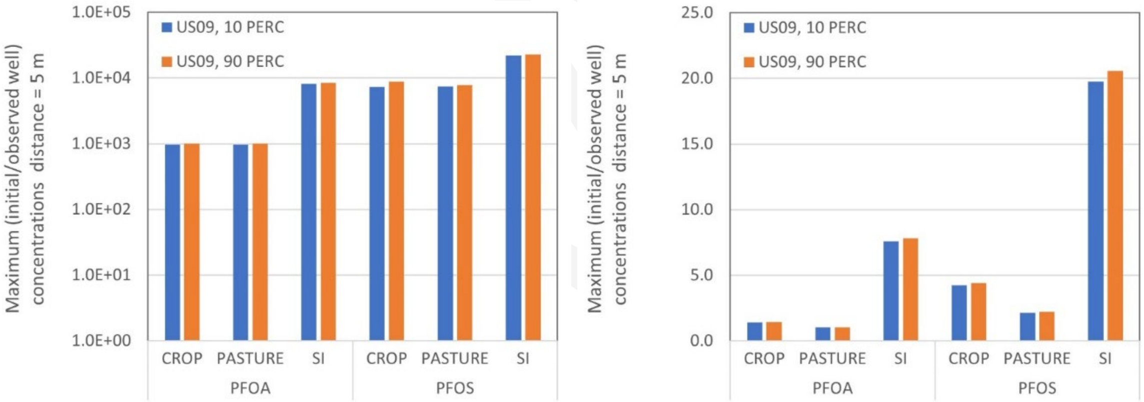


Figure D-8. Sensitivity to bulk density of the unsaturated zone for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

Effective porosity is the ratio of the volume of interconnected void spaces in rock or sediment to the total volume of rock or sediment. When not specified, the porosity, whether directly input or derived, is used to calculate the bulk density of the aquifer materials. The difference between the ratio of initial PFOA and PFOS source concentrations in biosolids leachate to predicted PFOA and PFOS concentrations at the observation well at the lower and upper bound values of this input parameter is less than a factor of 2 across PFOA and PFOS for all three scenarios and two locations (**Figure D-9**). Therefore, this input parameter is not considered to be sensitive to the predicted model risk outputs.

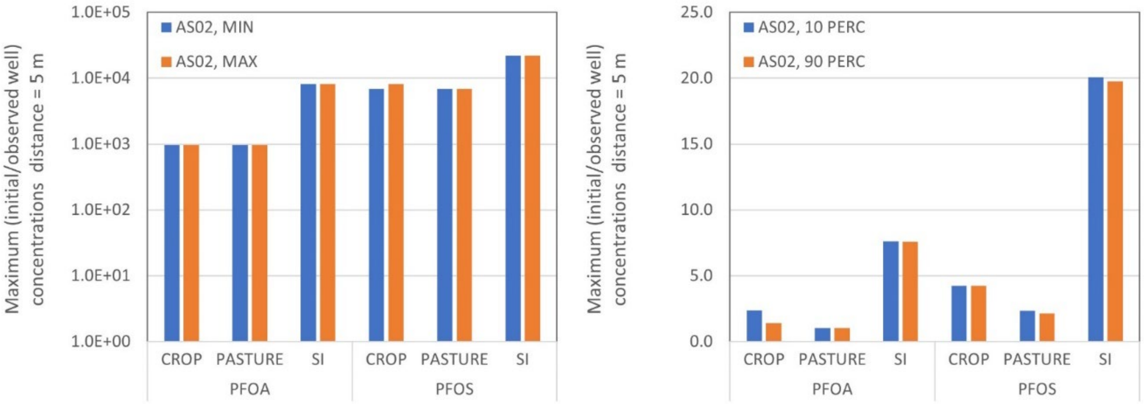


Figure D-9. Sensitivity to effective porosity of the unsaturated zone for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

D.3.1.2 Sensitivity to Saturated Zone Parameters

Bulk density is defined as the mass of aquifer solid material per unit volume of the aquifer, in g/cm3 or mg/L. Bulk density considers the fraction of the volume that is taken up by pore space. Bulk density is an input to the saturated zone flow and transport modules. In the saturated zone flow module, bulk density is used in the calculation of the ground-water seepage velocity. In the transport module, bulk density is one of several parameters used to calculate the degree to which contaminant velocities are retarded relative to the ambient groundwater flow velocity within the aquifer. The difference between the ratio of initial PFOA and PFOS source concentrations in biosolids leachate to predicted PFOA and PFOS concentrations at the observation well at the lower and upper bound values of this input parameter is less than a factor of 1.4 across PFOA and PFOS for all three scenarios and two locations (**Figure D-10**). Therefore, this input parameter is not considered to be sensitive to the predicted model risk outputs.

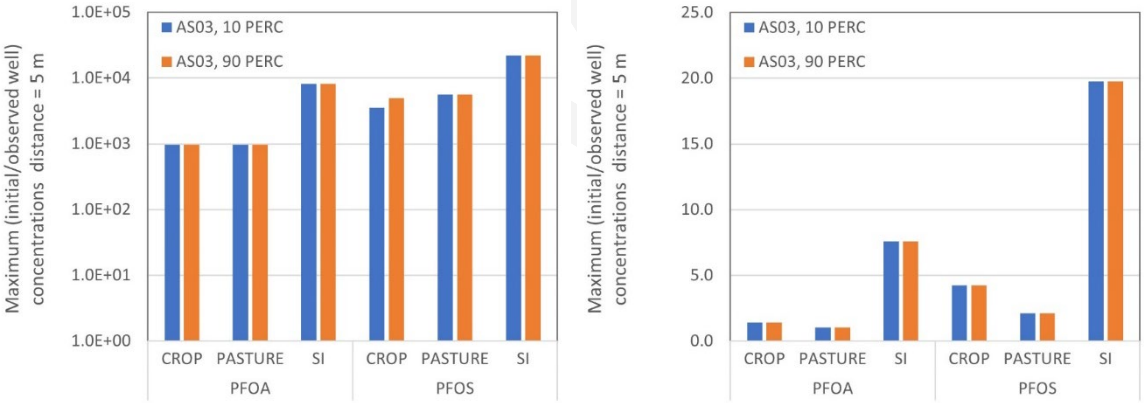
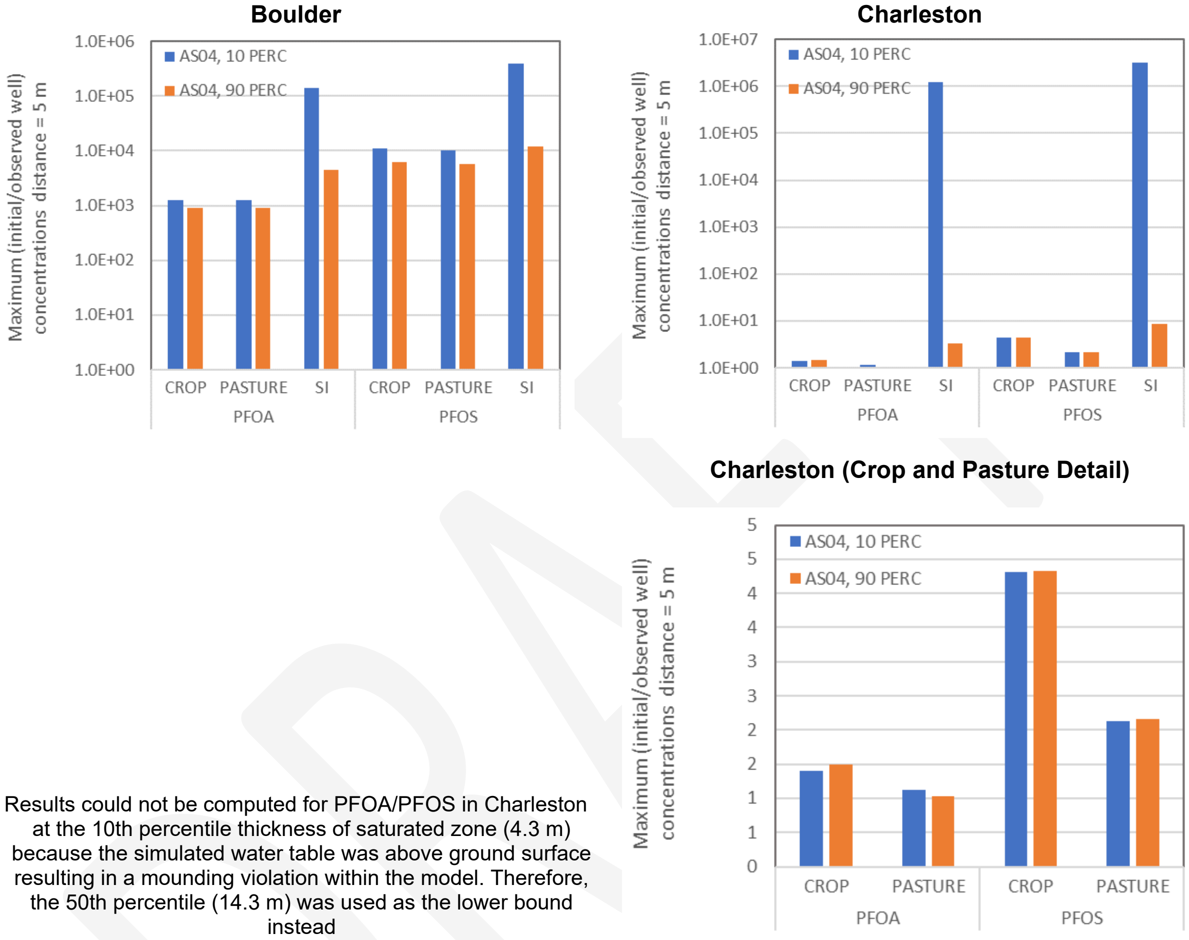


Figure D-10. Sensitivity to bulk density of the saturated zone for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

The **saturated zone thickness** is the vertical thickness of the zone in which the voids in the rock or soil are filled with water at a pressure greater than atmospheric. The thickness of the saturated zone is an input to the saturated zone flow module. It is used in EPACMTP to describe the thickness of the ground-water zone over which the leachate plume can mix with ground water and impacts the dilution rates in the saturated zone. The difference between the ratio of initial PFOA and PFOS source concentrations in biosolids leachate to predicted PFOA and PFOS concentrations at the observation well at the lower and upper bound values of this input parameter is a factor of several orders of magnitude across PFOA and

PFOS for all three scenarios and two locations (**Figure D-11**). This behavior is more pronounced for SI results due to the shallower penetration of dissolved contaminant into the aquifer from a source area that is more than 50 times smaller than the agricultural field – there is more attenuation at shallower depths that tends to increase the concentration ratios. Therefore, this input parameter is sensitive to the predicted model risk outputs.



Results could not be computed for PFOA/PFOS in Charleston at the 10th percentile thickness of saturated zone (4.3 m) because the simulated water table was above ground surface resulting in a mounding violation within the model. Therefore, the 50th percentile (14.3 m) was used as the lower bound instead

Figure D-11. Sensitivity to saturated zone thickness for Boulder (dry climate, left) and Charleston, SC (wet climate, right; top row on right shows LAU crop and pasture and SI, lower right shows detail for crop and pasture on a smaller y-axis scale).

Hydraulic conductivity is a measure of the ability to transmit water under a unit hydraulic gradient. The aquifer hydraulic conductivity is an input to the saturated zone flow module. The hydraulic conductivity, together with the hydraulic gradient, controls the ground-water flow rate. The difference between the ratio of initial PFOA and PFOS source concentrations in biosolids leachate to predicted PFOA and PFOS concentrations at the observation well at the lower and upper bound values of this input parameter is a factor of several orders of magnitude across PFOA and PFOS, particularly for SI scenarios at both geographic locations (**Figure D-12**). Therefore, this input parameter is sensitive to the predicted model risk outputs.

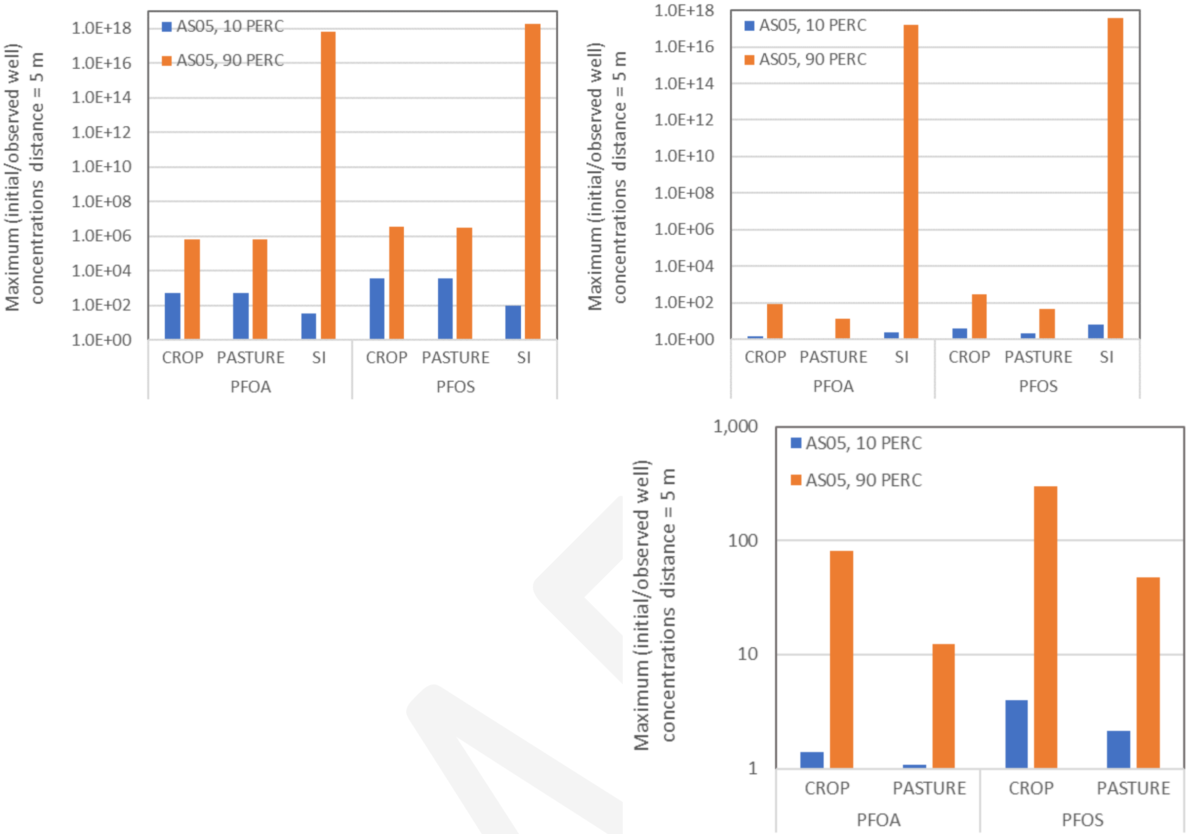


Figure D-12. Sensitivity to hydraulic conductivity of the saturated zone for Boulder (dry climate, left) and Charleston, SC (wet climate, right; top shows LAU crop and pasture and SI, lower shows detail for crop and pasture on a smaller y-axis scale).

Hydraulic gradient measures the head difference between two points as a function of their distance. For an unconfined aquifer such as that modeled with EPACMTP, the hydraulic gradient is simply the slope of the water table in a particular direction. It is calculated as the difference in the elevation of the water table measured at two locations divided by the distance between the two locations. The difference between the ratio of initial PFOA and PFOS source concentrations in biosolids leachate to predicted PFOA and PFOS concentrations at the observation well at the lower and upper bound values of this input parameter is a factor of several orders of magnitude across PFOA and PFOS, particularly for SI scenarios at both geographic locations (**Figure D-13**). Therefore, this input parameter is sensitive to the predicted model risk outputs.

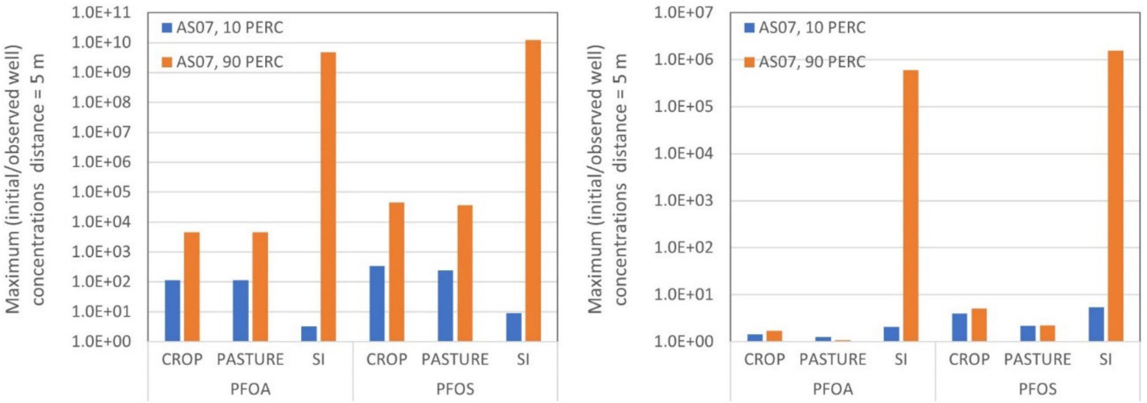


Figure D-13. Sensitivity to regional hydraulic gradient of the saturated zone for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

D.3.1.3 Sensitivity to Chemical-specific Parameter

The organic carbon partition coefficient (cm^3/g) is the ratio of a constituent's concentration in a theoretical soil containing only organic carbon to its concentration in the ground water. Thus, k$_{oc}$ describes the affinity of a constituent to attach itself to organic carbon. This parameter is applicable to organic constituents which tend to sorb onto the organic matter in soil or in an aquifer.

The figures in this section present a comparison of the ratio of initial PFOA and PFOS source concentrations in biosolids leachate to predicted PFOA and PFOS concentrations at an observation well located in the center of the buffer for crop, pasture and surface impoundment scenarios at three locations: Boulder, Chicago, and Charleston representing a dry, moderate and wet meteorology. All sensitivity simulations were performed assuming both the “low-Koc”, and “high-Koc” input parameters.

At all three locations, the ratio of initial PFOA and PFOS source concentrations in biosolids leachate to predicted PFOA and PFOS concentrations at an observation well located in the center of the buffer increases with an increase in the assumed K$_{oc}$ value. As noted earlier (**Section D.2.1**), for PFOS, the “low-K$_{oc}$” value of 2206.73 cm^3/g is greater than the “representative-K$_{oc}$” value of 371.5 cm^3/g. These results are expected as constituents with high Koc values tend to move more slowly through the soil and ground water. The effect of equilibrium sorption is expressed in EPACMTP through the retardation coefficient, R, which is a function of the chemical-specific organic carbon partition coefficient, k$_{oc}$:

$$R = 1 + \frac{\rho_b\, k_d}{\theta}$$

where,

R = retardation coefficient
ρ_b = bulk density of unsaturated soil (g/cm^3)
θ = soil water content (dimensionless)
k_d = soil-water partition coefficient (L/kg) = f_{oc} x k_{oc}
f_{oc} = fractional organic carbon content in the soil or aquifer (unitless)
k_{oc} = organic carbon partition coefficient (cm^3/g)

For a similar geographic location, representative scenario and K$_{oc}$ value tested, the ratio of initial PFOS source concentrations in biosolids leachate to predicted PFOS concentrations at an observation well located in the center of the buffer was observed to be greater than that for PFOA. For example,

Figure D-14 demonstrates this observation at Chicago for Crop and Pasture scenarios under low and high assumed $K_{oc.}$ This is consistent with a higher K_{oc} value for PFOS in comparison to PFOA, all environmental parameters being the same. A higher K_{oc} value for PFOS results in greater retardation and thus lower concentrations in the observation well located in the center of the buffer.

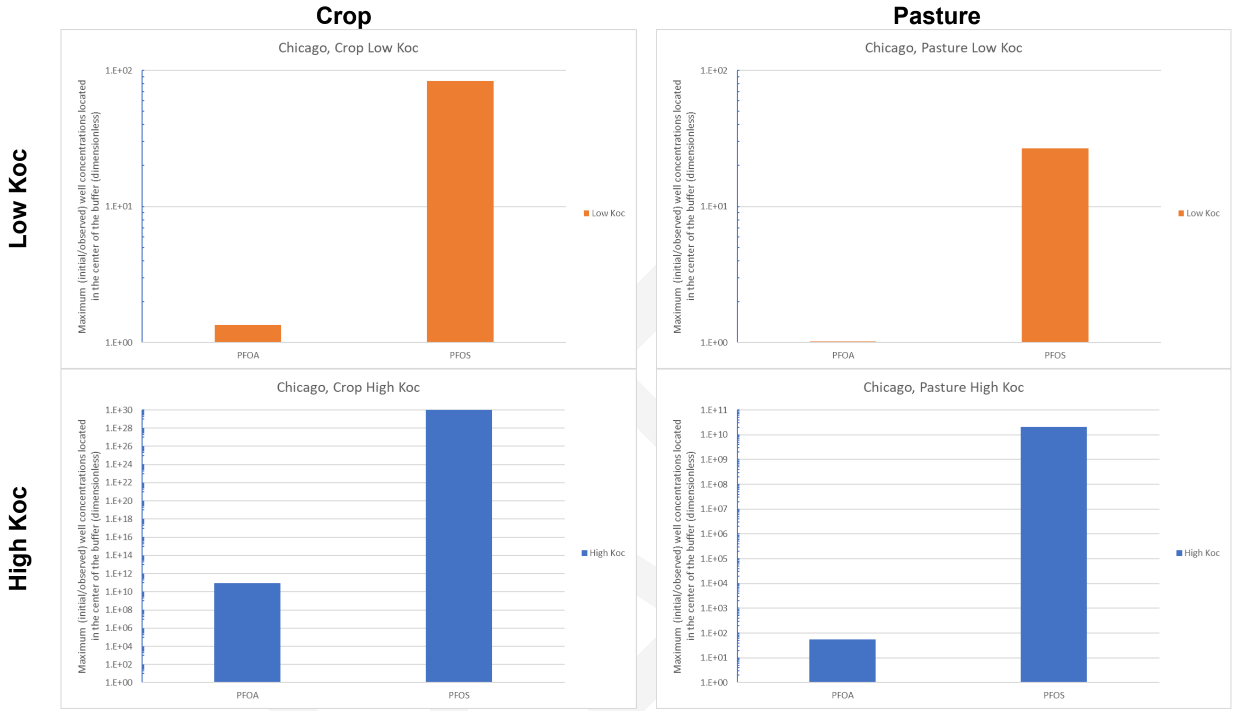


Figure D-14. Maximum leachate to well concentration ratio for land application unit for Chicago (moderate climate): crop (left) and pasture (right), low Koc (top) and high Koc (bottom).

The ratio of initial PFOA and PFOS source concentrations in biosolids leachate to predicted PFOA and PFOS concentrations at an observation well located in the center of the buffer was simulated to be different between geographic locations and corresponding meteorological conditions. Particularly, for the same representative scenario, chemical and K_{oc} value tested, higher PFOA and PFOS concentration ratios were observed at dry (Boulder) vs. wet (Charleston) conditions (**Figure D-15**). This suggests that lower PFOA and PFOS concentrations are observed at the well located in the center of the buffer under drier conditions likely due to lower mass transport on account of lesser infiltration.

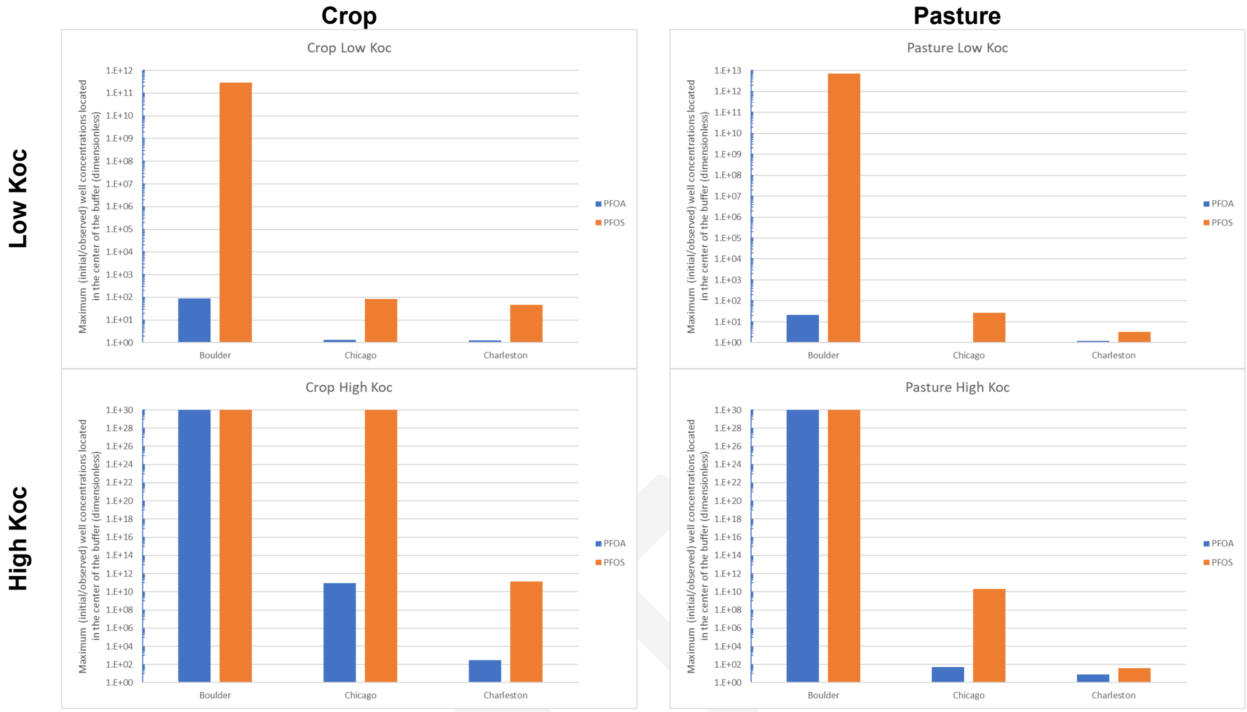


Figure D-15. Maximum leachate to well concentration ratio for land application unit by climate: crop (left) and pasture (right), low Koc (top) and high Koc (bottom).

The sensitivity of the ratio of initial PFOA and PFOS source concentrations in biosolids leachate to predicted PFOA and PFOS concentrations at an observation well located in the center of the buffer was also simulated for the surface impoundment scenario at all three geographic locations assuming a low- and high-Koc value (**Figure D-16**). Based on these simulations, higher concentration ratios are observed for a higher Koc value and more impermeable liner type (e.g., composite liner vs. clay liner), consistent with an expected lower mass transport.

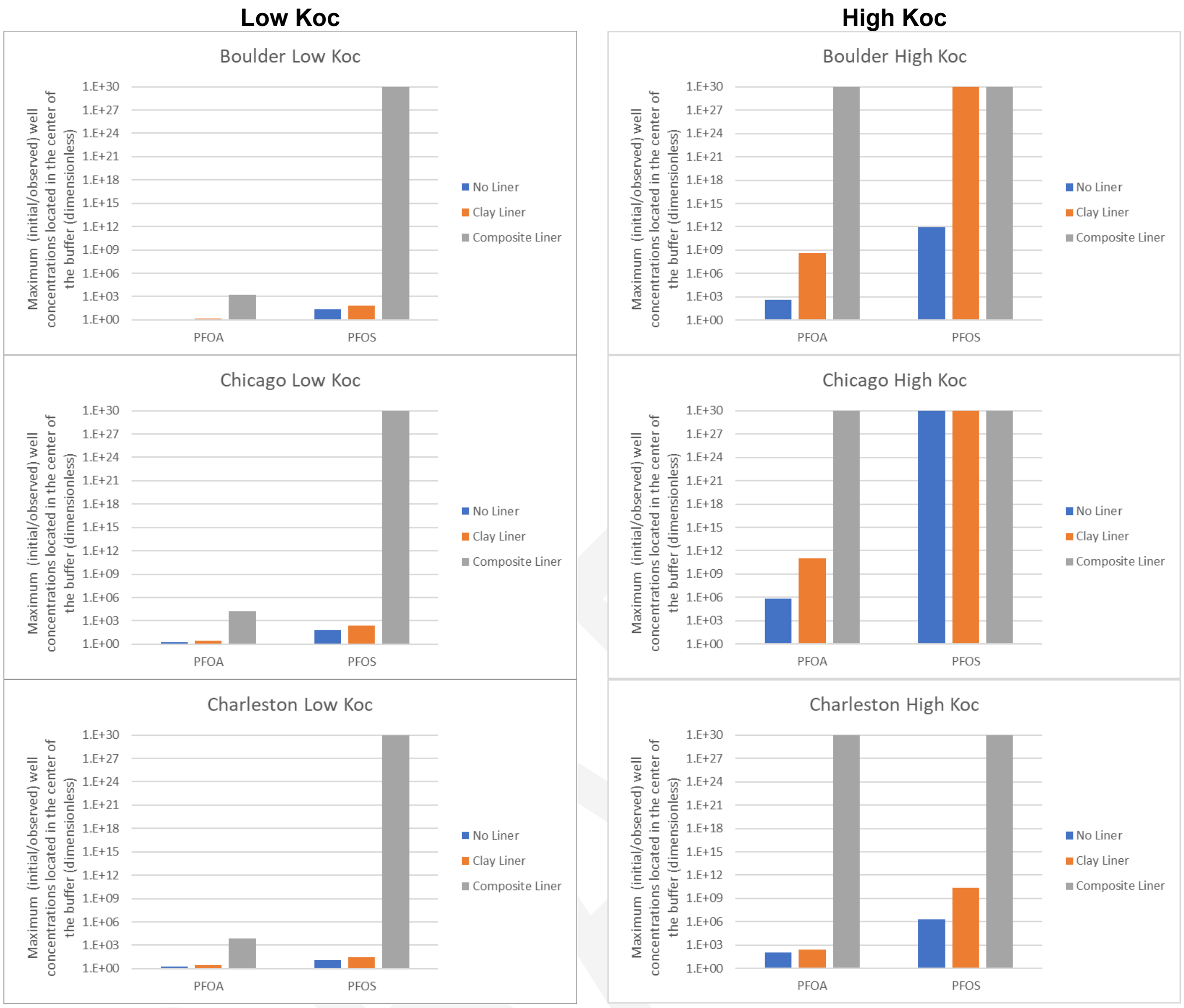


Figure D-16. Maximum leachate to well concentration ratio for surface disposal unit for low Koc (left column) and high Koc (right column) for Boulder (dry climate, top row), Chicago (moderate climate, middle row), and Charleston, SC (wet climate, bottom row).

Based on these observations, it can be concluded that the Koc, organic carbon partition coefficient, is a very sensitive parameter for all scenarios simulated.

D.3.1.4 Summary of Most Sensitive Parameters for EPACMTP

The parameters to which EPACMTP is most sensitive, defined as a change in well concentration of greater than a factor of 10 between the 10th percentile input value and the 90th percentile input value, are as follows:

- Depth from ground surface to water table (US06)
- Regional hydraulic gradient in the aquifer (AS07)
- Hydraulic conductivity of saturated zone (aquifer) (AS05)
- Saturated zone thickness (AS04)
- Organic carbon partition coefficient (Koc).

D.3.2 VVWM

Each bounding parameter value in Table D-3 was evaluated for both crop and pasture scenarios under dry and wet conditions for both PFOA and PFOS. A one-at-a-time approach was used to evaluate a bounding value for one parameter for all combinations of biosolids application scenarios (e.g., crop or pasture), and meteorological environments (e.g., dry or wet). All other parameters are represented by VVWM defaults or representative K_{oc} values. Peak concentration values corresponding to the adult receptor, surface water pathway, and noncancer benchmarks are used to calculate ratios of concentration corresponding to the bounding value of a parameter to the VVWM default value for the same parameter. To evaluate the sensitivity of flow through the surface water body, ratios of peak concentrations derived from dry and wet meteorology are examined for each chemical and application scenario. Chemical-related sensitivity was examined by computing ratios of peak concentration based on bounding values of K_{oc} case relative to default K_{oc} values for all combinations of dry and wet meteorology and application scenario.

D.3.2.1 Sensitivity to Media Parameters

Figure D-17 and **Figure D-18** present sensitivity results of the ratio of predicted peak, dissolved PFOA and PFOS surface water concentrations, respectively, using bounding parameter values to the same surface water concentrations simulated using default parameter values for Crop, Pasture scenarios in (a) Boulder [dry] and (b) Charleston [wet]. Ratios that are approximately 1.0 indicate that there was little or no difference in peak concentrations for result using a bounding value and baseline value. Ratios less than 1.0 indicate that the peak concentration simulated using the bounding parameter value was less than peak surface water concentration predicted using the baseline value for the same parameter. Likewise, rations greater than 1.0 indicate that the simulated peak concentration using the bounding value is greater than the corresponding simulation results for the baseline value for the same parameter. The only parameter showing any significant sensitivity is the fraction of organic carbon of the benthic sediments. Lower levels of organic carbon result in less sorption of a chemical to the sediments and higher dissolved concentrations and the converse for high FOC content. The difference in behaviors exhibited by the crop and pasture scenarios reflect the impact of application practices: tilling binds more mass to soil reducing concentrations in runoff and reducing the partitioning from eroded sediments making it to the reservoir where FOC is high. When FOC is low, more dissolved mass is moved off the field in runoff and released from solids reaching the reservoir.

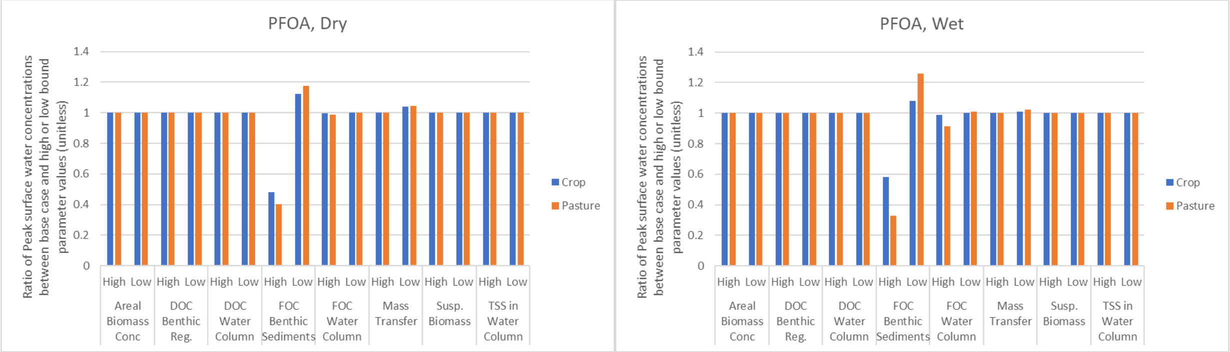


Figure D-17. Sensitivity of VVWM to media parameters for PFOA for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

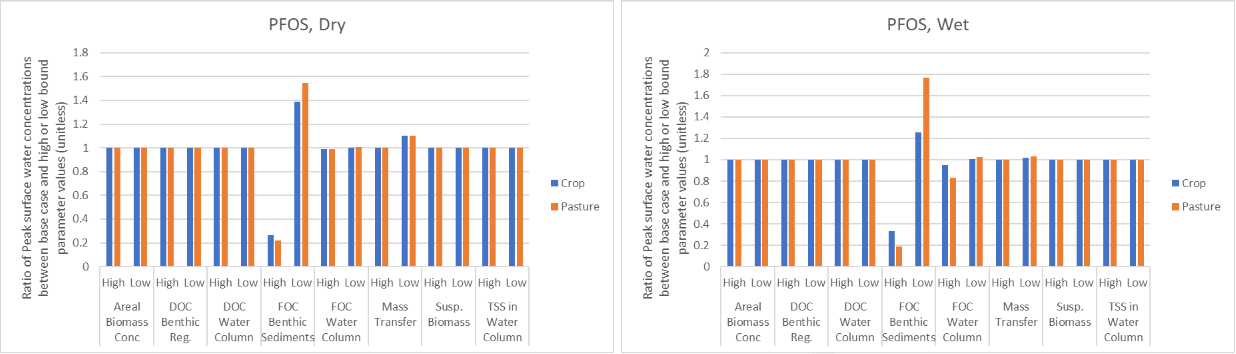


Figure D-18. Sensitivity of VVWM to media parameters for PFOS for Boulder (dry climate, left) and Charleston, SC (wet climate, right).

D.3.2.2 Sensitivity to Meteorology Parameters

Figure D-19 presents sensitivity of surface water concentrations to meteorological data as ratios of resulting concentrations from overland flow into and through the water body for dry conditions over wet conditions. The amount of mass available for a given scenario is fixed. Adding more precipitation increases the dilution of dissolved chemical in runoff, decreasing the concentration of the chemical entering the water body. As a result, all ratios are greater than 1.0.

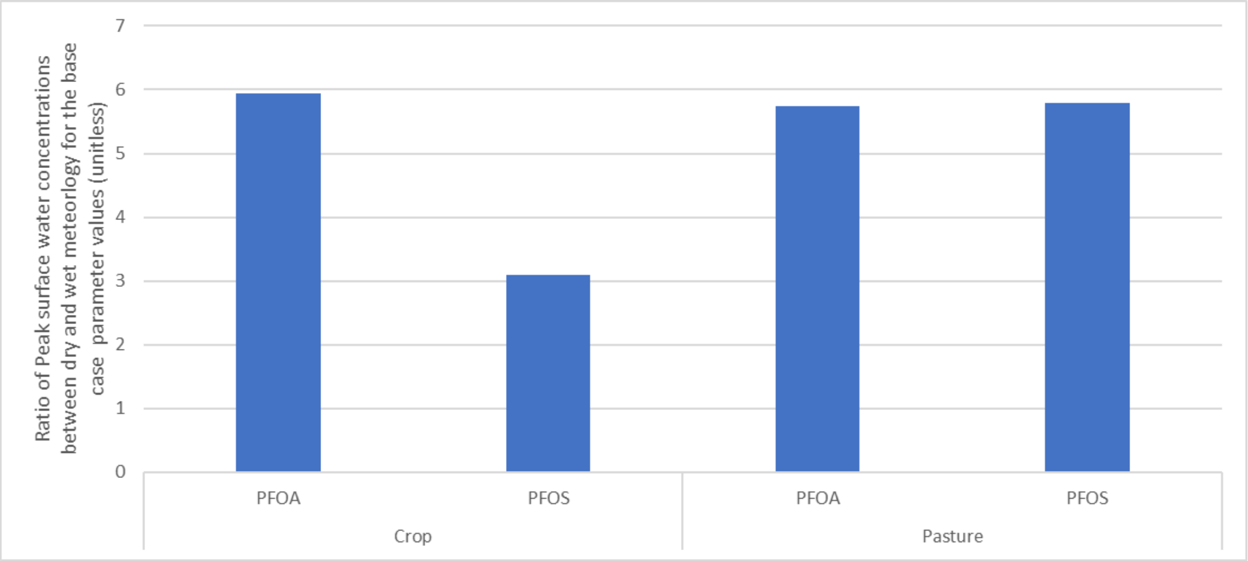


Figure D-19. Sensitivity of VVWM to meteorology for PFOA and PFOS for crop and pasture: ratio of peak surface water concentrations for dry climate to wet climate.

D.3.2.3 Sensitivity to Chemical Parameters

Figure D-20 presents sensitivity of surface water concentrations to low and high Koc values. Here, VVWM was run in isolation using a fixed loading and varying only K_{oc}. For PFOA and PFOS, VVWM is entirely insensitive to climate data; the results shown here are for Charleston (wet), but those for Boulder (dry) are identical. There is no distinction between crop and field here because that affects only the loading, not what happens within the surface water body. As noted earlier (**Section D.2.1**), for PFOS, the "low-K_{oc}" value of 2206.73 cm^3/g is greater than the "representative-K_{oc}" value of 371.5 cm^3/g.

As expected, as Koc increases, the fraction sorbed to bed sediment increases, increasing the concentration in sediments and decreasing the concentration in the water column.

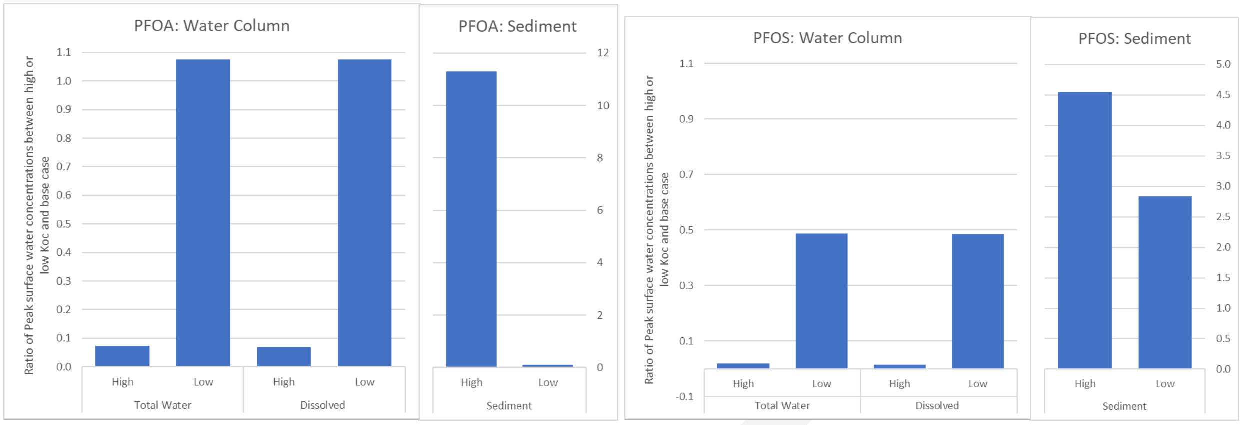


Figure D-20. Sensitivity of VVWM to Koc for PFOA (left) and PFOS (right).

D.3.2.4 Summary of Most Sensitive Parameters for VVWM

The parameters to which VVWM is most sensitive are as follows:

- Fraction of organic carbon of the benthic sediments
- Climate
- Organic carbon partition coefficient (Koc).

D.4 References

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APPENDIX E. SCREENING-LEVEL RESULTS FROM BST

E.1 BST Screening Inputs

Table E-1. Scenario Inputs

Scenario	Model_Code	Model Value	Units	Description	Reference
Crop	OpLife	40	[yrs]	Number of year of biosolids applications to field	Biosolids 2003 (US EPA 2003)
Crop	Nappl	1	[1/yr]	Number of biosolids applications per year	Biosolids 2003 (US EPA 2003)
Crop	Rappl	0.0025	[MTwet/m2-yr]	Biosolids application rate (wet weight)	Calculated
Crop	zruf	1	[cm]	Roughness height (field)	TSDF Fugit. Air (US EPA, 1989b)
Crop	Ztilling	0.2	[m]	Tilling depth	Biosolids 2003 (US EPA 2003)
Pasture	OpLife	40	[yrs]	Number of year of biosolids applications to field	Biosolids 2003 (US EPA 2003)
Pasture	Nappl	1	[1/yr]	Number of biosolids applications per year	Biosolids 2003 (US EPA 2003)
Pasture	Rappl	0.0025	[MTwet/m2-yr]	Biosolids application rate (wet weight)	Calculated
Pasture	zruf	3.7	[cm]	Roughness height (field)	TSDF Fugit. Air (US EPA, 1989b)
Pasture	Ztilling	0.02	[m]	Tilling depth	Biosolids 2003 (US EPA 2003)
Reclamation	OpLife	1	[yrs]	Number of year of biosolids applications to field	Biosolids 2003 (US EPA 2003)
Reclamation	Nappl	1	[1/yr]	Number of biosolids applications per year	Biosolids 2003 (US EPA 2003)
Reclamation	Rappl	0.0125	[MTwet/m2-yr]	Biosolids application rate (wet weight)	Calculated
Reclamation	zruf	3.7	[cm]	Roughness height (field)	TSDF Fugit. Air (US EPA, 1989b)
Reclamation	Ztilling	0.02	[m]	Tilling depth	Biosolids 2003 (US EPA 2003)

Table E-2. Fate Inputs

Model_Code	Moderate Value	Dry Value	Wet Value	Units	Description	Reference	Comment
%solids	40	40	40	[mass %]	Percent solids in land applied biosolids	Biosolids 2003 (US EPA 2003)	
AirTemp	9.69	10.11	18.18	[C]	Average air temperature	SAMSON (US DOC & DOE, 1993)	determined by met station
Area_reserv	52555	52555	52555	[m2]	Area (index reservoir)	VVWM	
asdm	0.5	0.5	0.5	[mm]	Mode of the aggregate size distribution	TSDF Fugit. Air (US EPA, 1989b)	
Bdwaste	0.7	0.7	0.7	[g DW/cm3]	Dry bulk density (biosolids)	Gunn et al. (2004)	
bsp	0.6	0.6	0.6	[fraction]	Porosity (bed sediment)	MPE/IEM (US EPA, 1998)	
C	0.1	0.1	0.1	[fraction]	USLE cover management factor	HHRAP (US EPA, 2005a)	
db	0.05	0.05	0.05	[m]	Depth of upper benthic layer	MPE/IEM (US EPA, 1998)	changed for VVWM
DTR	12	12	12	[m2/m3]	Drainage area to capacity ratio (watershed)	SAB (Index Res)	
dwc_pond	2	2	2	[m]	Water column depth (farm pond)	VVWM	
dwc_reservoir	2.74	2.74	2.74	[m]	Water column depth (index reservoir)	VVWM	
foc_bedsed	0.04	0.04	0.04	[fraction]	Fraction organic carbon (bed sediments)	VVWM	
foc_biosolids	0.4	0.4	0.4	[fraction]	Fraction organic carbon (biosolids)	Biosolids 2003 (US EPA 2003)	
foc_sw	0.04	0.04	0.04	[fraction]	Fraction organic carbon (suspended sediments)	VVWM	
P	1	1	1	[fraction]	USLE supporting practice factor (watershed)	Wanielista & Yousef, 1993	

Model_Code	Moderate Value	Dry Value	Wet Value	Units	Description	Reference	Comment
PI_field	0	0	0	[%]	Percent impervious (field)	CWP, 1998	
R	155	50	360	[1/yr]	USLE rainfall/erosivity factor	Wischmeier & Smith, 1978	determined by met station
SiteLatitude	41.983	40.0167	32.9	[degrees]	Site latitude	SAMSON (US DOC & DOE, 1993)	determined by met station
Sw	10	10	10	[mass %]	Silt content of biosolids	AP-42 (US EPA, 1995)	
Theta_water	1.024	1.024	1.024	[empirical]	Temperature correction factor	Chapra, 1996	
Twater01	270	273	284	[deg K]	Waterbody temperature (January)	Water Enc. (van der Leeden et al., 1990)	depends on HUC Region
Twater02	267	271	282	[deg K]	Waterbody temperature (February)	Water Enc. (van der Leeden et al., 1990)	depends on HUC Region
Twater03	270	274	283	[deg K]	Waterbody temperature (March)	Water Enc. (van der Leeden et al., 1990)	depends on HUC Region
Twater04	276	277	287	[deg K]	Waterbody temperature (April)	Water Enc. (van der Leeden et al., 1990)	depends on HUC Region
Twater05	282	282	291	[deg K]	Waterbody temperature (May)	Water Enc. (van der Leeden et al., 1990)	depends on HUC Region
Twater06	289	287	295	[deg K]	Waterbody temperature (June)	Water Enc. (van der Leeden et al., 1990)	depends on HUC Region
Twater07	294	293	299	[deg K]	Waterbody temperature (July)	Water Enc. (van der Leeden et al., 1990)	depends on HUC Region
Twater08	297	296	300	[deg K]	Waterbody temperature (August)	Water Enc. (van der Leeden et al., 1990)	depends on HUC Region
Twater09	295	295	299	[deg K]	Waterbody temperature (September)	Water Enc. (van der Leeden et al., 1990)	depends on HUC Region
Twater10	291	290	297	[deg K]	Waterbody temperature (October)	Water Enc. (van der Leeden et al., 1990)	depends on HUC Region
Twater11	285	284	292	[deg K]	Waterbody temperature (November)	Water Enc. (van der Leeden et al., 1990)	depends on HUC Region
Twater12	278	277	288	[deg K]	Waterbody temperature (December)	Water Enc. (van der Leeden et al., 1990)	depends on HUC Region
uw	4.632	3.783	3.788	[m/sec]	Mean annual wind speed	SAMSON (US DOC & DOE, 1993)	determined by met station
zavg_lower	0.2	0.2	0.2	[m]	Lower averaging depth for soil concentration	Biosolids 2003 (US EPA 2003)	
Zmix_tilled	20	20	20	[cm]	Mixing depth of tilled soil (field)	Biosolids 2003 (US EPA 2003)	
Zmix_untilled	2	2	2	[cm]	Mixing depth of untilled soil (field)	Biosolids 2003 (US EPA 2003)	
Zmodeled	0.2	0.2	0.2	[m]	Depth of modeled soil column	Biosolids 2003 (US EPA 2003)	

Table E-3. Exposure Inputs

Scenario	Receptor	Model_Code	Model Value	Units	Description	Reference
Crop	Farmer	BW_child1-5	15	[kg]	Body weight (child aged 1-5)	EFH:2011 (US EPA, 2011)
Crop	Farmer	BW_child6-11	29	[kg]	Body weight (child aged 6-11)	EFH:2011 (US EPA, 2011)
Crop	Farmer	BW_child12-19	61	[kg]	Body weight (child aged 12-19)	EFH:2011 (US EPA, 2011)
Crop	Farmer	BW_adult	79	[kg]	Body weight (adult)	EFH:2011 (US EPA, 2011)
Crop	Farmer	CR_dw_child1-5	44	[mL/kg-day]	Consumption rate, water (child aged 1-5)	EFH:2019-dw
Crop	Farmer	CR_dw_child6-11	31	[mL/kg-day]	Consumption rate, water (child aged 6-11)	EFH:2019-dw
Crop	Farmer	CR_dw_child12-19	25	[mL/kg-day]	Consumption rate, water (child aged 12-19)	EFH:2019-dw
Crop	Farmer	CR_dw_adult	34	[mL/kg-day]	Consumption rate, water (adult)	EFH:2019-dw
Crop	Farmer	CR_exfruit_child1-5	5.4	[g WW/kg BW/day]	Consumption rate, exposed fruit (child aged 1-5)	EFH:2011 (US EPA, 2011)
Crop	Farmer	CR_exfruit_child6-11	7	[g WW/kg BW/day]	Consumption rate, exposed fruit (child aged 6-11)	EFH:2011 (US EPA, 2011)
Crop	Farmer	CR_exfruit_child12-19	3.4	[g WW/kg BW/day]	Consumption rate, exposed fruit (child aged 12-19)	EFH:2011 (US EPA, 2011)
Crop	Farmer	CR_exfruit_adult	5	[g WW/kg BW/day]	Consumption rate, exposed fruit (adult)	EFH:2011 (US EPA, 2011)
Crop	Farmer	CR_exveg_child1-5	6.4	[g WW/kg BW/day]	Consumption rate, exposed vegetables (child aged 1-5)	EFH:2011 (US EPA, 2011)
Crop	Farmer	CR_exveg_child6-11	3.2	[g WW/kg BW/day]	Consumption rate, exposed vegetables (child aged 6-11)	EFH:2011 (US EPA, 2011)
Crop	Farmer	CR_exveg_child12-19	2.4	[g WW/kg BW/day]	Consumption rate, exposed vegetables (child aged 12-19)	EFH:2011 (US EPA, 2011)

Scenario	Receptor	Model_Code	Model Value	Units	Description	Reference
Crop	Farmer	CR_exveg_adult	6	[g WW/kg BW/day]	Consumption rate, exposed vegetables (adult)	EFH:2011 (US EPA, 2011)
Crop	Farmer	CR_fish_child1-5	5.2	[g WW/day]	Consumption rate, fish (child aged 1-5)	NHANES:2014-fish
Crop	Farmer	CR_fish_child6-11	7.7	[g WW/day]	Consumption rate, fish (child aged 6-11)	NHANES:2014-fish
Crop	Farmer	CR_fish_child12-19	9.6	[g WW/day]	Consumption rate, fish (child aged 12-19)	NHANES:2014-fish
Crop	Farmer	CR_fish_adult	22	[g WW/day]	Consumption rate, fish (adult)	NHANES:2014-fish
Crop	Farmer	CR_profruit_child1-5	16	[g WW/kg BW/day]	Consumption rate, protected fruit (child aged 1-5)	EFH:2011 (US EPA, 2011)
Crop	Farmer	CR_profruit_child6-11	16	[g WW/kg BW/day]	Consumption rate, protected fruit (child aged 6-11)	EFH:2011 (US EPA, 2011)
Crop	Farmer	CR_profruit_child12-19	7.4	[g WW/kg BW/day]	Consumption rate, protected fruit (child aged 12-19)	EFH:2011 (US EPA, 2011)
Crop	Farmer	CR_profruit_adult	14	[g WW/kg BW/day]	Consumption rate, protected fruit (adult)	EFH:2011 (US EPA, 2011)
Crop	Farmer	CR_proveg_child1-5	3.1	[g WW/kg BW/day]	Consumption rate, protected vegetables (child aged 1-5)	EFH:2011 (US EPA, 2011)
Crop	Farmer	CR_proveg_child6-11	2.1	[g WW/kg BW/day]	Consumption rate, protected vegetables (child aged 6-11)	EFH:2011 (US EPA, 2011)
Crop	Farmer	CR_proveg_child12-19	1.9	[g WW/kg BW/day]	Consumption rate, protected vegetables (child aged 12-19)	EFH:2011 (US EPA, 2011)
Crop	Farmer	CR_proveg_adult	3.6	[g WW/kg BW/day]	Consumption rate, protected vegetables (adult)	EFH:2011 (US EPA, 2011)
Crop	Farmer	CR_root_child1-5	5.7	[g WW/kg BW/day]	Consumption rate, root vegetables (child aged 1-5)	EFH:2011 (US EPA, 2011)
Crop	Farmer	CR_root_child6-11	3.8	[g WW/kg BW/day]	Consumption rate, root vegetables (child aged 6-11)	EFH:2011 (US EPA, 2011)
Crop	Farmer	CR_root_child12-19	2.3	[g WW/kg BW/day]	Consumption rate, root vegetables (child aged 12-19)	EFH:2011 (US EPA, 2011)
Crop	Farmer	CR_root_adult	3.1	[g WW/kg BW/day]	Consumption rate, root vegetables (adult)	EFH:2011 (US EPA, 2011)
Crop	Farmer	CRs_child1-5	40	[mg/day]	Consumption rate, soil (child aged 1-5)	EFH:2017-soil
Crop	Farmer	CRs_child6-11	30	[mg/day]	Consumption rate, soil (child aged 6-11)	EFH:2017-soil
Crop	Farmer	CRs_child12-19	10	[mg/day]	Consumption rate, soil (child aged 12-19)	EFH:2017-soil
Crop	Farmer	CRs_adult	10	[mg/day]	Consumption rate, soil (adult)	EFH:2017-soil
Crop	Farmer	ED_child1-5	13	[yrs]	Exposure duration (child aged 1-5)	EFH:2011 (US EPA, 2011)
Crop	Farmer	ED_child6-11	13	[yrs]	Exposure duration (child aged 6-11)	EFH:2011 (US EPA, 2011)
Crop	Farmer	ED_child12-19	13	[yrs]	Exposure duration (child aged 12-19)	EFH:2011 (US EPA, 2011)
Crop	Farmer	ED_adult	48	[yrs]	Exposure duration (adult)	EFH:2011 (US EPA, 2011)
Crop	Farmer	ShowerTime	15	[min]	Time in shower stall during shower	EFH:2011 (US EPA, 2011)
Crop	Farmer	T_bathroom	5	[min]	Time spent in bathroom, not in shower	EFH:2011 (US EPA, 2011)
Pasture	Farmer	BW_child1-5	15	[kg]	Body weight (child aged 1-5)	EFH:2011 (US EPA, 2011)
Pasture	Farmer	BW_child6-11	29	[kg]	Body weight (child aged 6-11)	EFH:2011 (US EPA, 2011)
Pasture	Farmer	BW_child12-19	61	[kg]	Body weight (child aged 12-19)	EFH:2011 (US EPA, 2011)
Pasture	Farmer	BW_adult	79	[kg]	Body weight (adult)	EFH:2011 (US EPA, 2011)
Pasture	Farmer	CR_beef_child1-5	11	[g WW/kg BW/day]	Consumption rate, beef (child aged 1-5)	EFH:2011 (US EPA, 2011)
Pasture	Farmer	CR_beef_child6-11	11	[g WW/kg BW/day]	Consumption rate, beef (child aged 6-11)	EFH:2011 (US EPA, 2011)
Pasture	Farmer	CR_beef_child12-19	3.5	[g WW/kg BW/day]	Consumption rate, beef (child aged 12-19)	EFH:2011 (US EPA, 2011)
Pasture	Farmer	CR_beef_adult	5.4	[g WW/kg BW/day]	Consumption rate, beef (adult)	EFH:2011 (US EPA, 2011)
Pasture	Farmer	CR_dw_child1-5	44	[mL/kg-day]	Consumption rate, water (child aged 1-5)	EFH:2019-dw
Pasture	Farmer	CR_dw_child6-11	31	[mL/kg-day]	Consumption rate, water (child aged 6-11)	EFH:2019-dw
Pasture	Farmer	CR_dw_child12-19	25	[mL/kg-day]	Consumption rate, water (child aged 12-19)	EFH:2019-dw
Pasture	Farmer	CR_dw_adult	34	[mL/kg-day]	Consumption rate, water (adult)	EFH:2019-dw
Pasture	Farmer	CR_milk_child1-5	59	[g WW/kg BW/day]	Consumption rate, milk (child aged 1-5)	EFH:2018-meatdairy
Pasture	Farmer	CR_milk_child6-11	26	[g WW/kg BW/day]	Consumption rate, milk (child aged 6-11)	EFH:2018-meatdairy
Pasture	Farmer	CR_milk_child12-19	12	[g WW/kg BW/day]	Consumption rate, milk (child aged 12-19)	EFH:2018-meatdairy

Scenario	Receptor	Model_Code	Model Value	Units	Description	Reference
Pasture	Farmer	CR_milk_adult	35	[g WW/kg BW/day]	Consumption rate, milk (adult)	EFH:2011 (US EPA, 2011)
Pasture	Farmer	ED_child1-5	13	[yrs]	Exposure duration (child aged 1-5)	EFH:2011 (US EPA, 2011)
Pasture	Farmer	ED_child6-11	13	[yrs]	Exposure duration (child aged 6-11)	EFH:2011 (US EPA, 2011)
Pasture	Farmer	ED_child12-19	13	[yrs]	Exposure duration (child aged 12-19)	EFH:2011 (US EPA, 2011)
Pasture	Farmer	ED_adult	48	[yrs]	Exposure duration (adult)	EFH:2011 (US EPA, 2011)
Pasture	Farmer	ShowerTime	15	[min]	Time in shower stall during shower	EFH:2011 (US EPA, 2011)
Pasture	Farmer	T_bathroom	5	[min]	Time spent in bathroom, not in shower	EFH:2011 (US EPA, 2011)
Reclamation	Farmer	BW_child1-5	15	[kg]	Body weight (child aged 1-5)	EFH:2011 (US EPA, 2011)
Reclamation	Farmer	BW_child6-11	29	[kg]	Body weight (child aged 6-11)	EFH:2011 (US EPA, 2011)
Reclamation	Farmer	BW_child12-19	61	[kg]	Body weight (child aged 12-19)	EFH:2011 (US EPA, 2011)
Reclamation	Farmer	BW_adult	79	[kg]	Body weight (adult)	EFH:2011 (US EPA, 2011)
Reclamation	Farmer	CR_beef_child1-5	11	[g WW/kg BW/day]	Consumption rate, beef (child aged 1-5)	EFH:2011 (US EPA, 2011)
Reclamation	Farmer	CR_beef_child6-11	11	[g WW/kg BW/day]	Consumption rate, beef (child aged 6-11)	EFH:2011 (US EPA, 2011)
Reclamation	Farmer	CR_beef_child12-19	3.5	[g WW/kg BW/day]	Consumption rate, beef (child aged 12-19)	EFH:2011 (US EPA, 2011)
Reclamation	Farmer	CR_beef_adult	5.4	[g WW/kg BW/day]	Consumption rate, beef (adult)	EFH:2011 (US EPA, 2011)
Reclamation	Farmer	CR_dw_child1-5	44	[mL/kg-day]	Consumption rate, water (child aged 1-5)	EFH:2019-dw
Reclamation	Farmer	CR_dw_child6-11	31	[mL/kg-day]	Consumption rate, water (child aged 6-11)	EFH:2019-dw
Reclamation	Farmer	CR_dw_child12-19	25	[mL/kg-day]	Consumption rate, water (child aged 12-19)	EFH:2019-dw
Reclamation	Farmer	CR_dw_adult	34	[mL/kg-day]	Consumption rate, water (adult)	EFH:2019-dw
Reclamation	Farmer	CR_milk_child1-5	59	[g WW/kg BW/day]	Consumption rate, milk (child aged 1-5)	EFH:2018-meatdairy
Reclamation	Farmer	CR_milk_child6-11	26	[g WW/kg BW/day]	Consumption rate, milk (child aged 6-11)	EFH:2018-meatdairy
Reclamation	Farmer	CR_milk_child12-19	12	[g WW/kg BW/day]	Consumption rate, milk (child aged 12-19)	EFH:2018-meatdairy
Reclamation	Farmer	CR_milk_adult	35	[g WW/kg BW/day]	Consumption rate, milk (adult)	EFH:2011 (US EPA, 2011)
Reclamation	Farmer	ED_child1-5	13	[yrs]	Exposure duration (child aged 1-5)	EFH:2011 (US EPA, 2011)
Reclamation	Farmer	ED_child6-11	13	[yrs]	Exposure duration (child aged 6-11)	EFH:2011 (US EPA, 2011)
Reclamation	Farmer	ED_child12-19	13	[yrs]	Exposure duration (child aged 12-19)	EFH:2011 (US EPA, 2011)
Reclamation	Farmer	ED_adult	48	[yrs]	Exposure duration (adult)	EFH:2011 (US EPA, 2011)
Reclamation	Farmer	ShowerTime	15	[min]	Time in shower stall during shower	EFH:2011 (US EPA, 2011)
Reclamation	Farmer	T_bathroom	5	[min]	Time spent in bathroom, not in shower	EFH:2011 (US EPA, 2011)

Table E-4. Chemical-specific Inputs

Chemical Name	Model_Code	Value	Units	User Modified	Description	Reference	UserComment
PFOA	BCF_beef	0.153	[mg/kg beef]/[mg/kg DW]	FALSE	Bioconcentration factor (beef)	Vestergren et al. (2013)	
PFOA	BCF_milk	0.233	[mg/kg milk]/[mg/kg DW]	FALSE	Bioconcentration factor (milk)	Vestergren et al. (2013)	
PFOA	BCF_T3F	8.5	[mg/kg fish]/[mg/L water]	FALSE	Bioaccumulation factor (TL3 fish, filet; used for human)	Burkhard 2021	Table 4
PFOA	BCF_T3W	140	[mg/kg fish]/[mg/L water]	FALSE	Bioaccumulation factor (TL3 fish, whole; used for eco)	Burkhard 2021	Table 4

Chemical Name	Model_Code	Value	Units	User Modified	Description	Reference	UserComment
PFOA	BCF_T4F	8.5	[mg/kg fish]/[mg/L water]	FALSE	Bioaccumulation factor (TL4 fish, filet; used for human)	Burkhard 2021	Table 4
PFOA	BCF_T4W	140	[mg/kg fish]/[mg/L water]	FALSE	Bioaccumulation factor (TL4 fish, whole; used for eco)	Burkhard 2021	Table 4
PFOA	BrExFruit	0.11	[mg/kg DW plant]/[mg/kg soil]	FALSE	Biotransfer (soil to exposed fruit)	Blaine et al. (2014)	
PFOA	BrExVeg	1.6	[mg/kg DW plant]/[mg/kg soil]	FALSE	Biotransfer factor (soil to exposed vegetables)	Blaine et al. (2013)	
PFOA	BrForage	0.25	[mg/kg DW plant]/[mg/kg soil]	FALSE	Biotransfer factor (soil to forage)	Blaine et al. (2013)	
PFOA	BrGrain	0.25	[mg/kg DW plant]/[mg/kg soil]	FALSE	Biotransfer factor (soil to grain)	Blaine et al. (2013)	
PFOA	BrProFruit	0.11	[mg/kg DW plant]/[mg/kg soil]	FALSE	Biotransfer factor (soil to protected fruit)	Blaine et al. (2014)	
PFOA	BrProVeg	1.6	[mg/kg DW plant]/[mg/kg soil]	FALSE	Biotransfer factor (soil to protected vegetables)	Blaine et al. (2013)	
PFOA	BrSilage	0.25	[mg/kg DW plant]/[mg/kg soil]	FALSE	Biotransfer factor (soil to silage)	Blaine et al. (2013)	
PFOA	Bv	0	[ug/g DW plant]/[ug/g air]	FALSE	Biotransfer factor (vapor phase air to all plants; organics only)	No Data	No data available
PFOA	CSFOral	29300	[per mg/kg-day]	FALSE	Oral cancer slope factor (human toxicity)	Prop PFOA-PFOS Tox	October 2022 Candidate Tox Values
PFOA	CTPWasteDry	0.307	[ug/g DW]	FALSE	Dry biosolids concentration	VT DEC	
PFOA	Da	0.0515	[cm2/s]	FALSE	Diffusivity in air	EPA Estimation Tool	
PFOA	Dw	5.52E-6	[cm2/s]	FALSE	Diffusion coefficient in water	EPA Estimation Tool	
PFOA	Heat_of_Henry	50000	[J/mol]	FALSE	Enthalpy of phase transformation from aqueous solution to air solution	EPISuite (US EPA, 2010)	
PFOA	HLC	0	[atm-m3/mol]	FALSE	Henry's law constant	HSDB (US NLM, 2010)	Sec 6.10, "not expected to volatilize from water or moist soil"
PFOA	IUR	0	[ug/m3]^-1	FALSE	Inhalation unit risk (human toxicity, cancer)	No Data	
PFOA	kaer	0	(1/day)	FALSE	Aerobic biodegradation rate (surface-water column)	No Data	No data, estimation tools not appropriate
PFOA	Kanaer	0	[1/day]	FALSE	Anaerobic degradation rate (sediment)	No Data	No data, estimation tools not appropriate
PFOA	kh	0	[1/day]	FALSE	Hydrolysis rate	No Data	No data, estimation tools not appropriate
PFOA	Koc	114.8	[mL/g]	FALSE	Organic carbon partition coefficient	PFOA Health Effects Support Document	

Chemical Name	Model_Code	Value	Units	User Modified	Description	Reference	UserComment
PFOA	Kpo	0	(1/day)	FALSE	Photolysis degradation rate in the surface of the water column	No Data	
PFOA	ksoil	0	[1/day]	FALSE	Biodegradation rate (soil)	No Data	No data, estimation tools not appropriate
PFOA	MW	414	[g/mol]	FALSE	Molecular weight	HSDB (US NLM, 2010)	
PFOA	RCF	0.02	[ug/g WW plant]/[ug/mL soil water]	FALSE	Root concentration factor*	Lechner and Knapp (2011)	
PFOA	Ref_BMD_Bird	0	[mg chem/kg BW/day]	FALSE	Reference benchmark dose (bird)	No Data	
PFOA	Ref_BMD_Mammal	0	[mg chem/kg BW/day]	FALSE	Reference benchmark dose (mammal)	No Data	
PFOA	Ref_BW_Bird	0	[kg]	FALSE	Reference body weight (bird)	No Data	
PFOA	Ref_BW_Mammal	0	[kg]	FALSE	Reference body weight (mammal)	No Data	
PFOA	RFC	0	[mg/m3]	FALSE	Reference concentration (human toxicity, noncancer)	No Data	
PFOA	RfD	3e-8	[mg/kg-day]	FALSE	Reference dose (human toxicity, noncancer)	Prop PFOA-PFOS Tox	October 2022 Candidate Tox Values
PFOA	RFD_By_Pathway	False	NA	FALSE	True if Model run with pathway-specific RfD	System	
PFOA	Sol	0.0095	[mg/L]	FALSE	Solubility	Physprop	
PFOA	temp_ref_aer_all	25	[C]	FALSE	Reference temperature for water column degradation	Default	
PFOA	temp_ref_anae_all	25	[C]	FALSE	Reference temperature for benthic degradation	Default	
PFOS	BCF_beef	0.874	[mg/kg beef]/[mg/kg DW]	FALSE	Bioconcentration factor (beef)	Vestergren et al. (2013)	
PFOS	BCF_milk	0.44	[mg/kg milk]/[mg/kg DW]	FALSE	Bioconcentration factor (milk)	Vestergren et al. (2013)	
PFOS	BCF_T3F	1500	[mg/kg fish]/[mg/L water]	FALSE	Bioaccumulation factor (TL3 fish, filet; used for human)	Burkhard 2021	Table 4
PFOS	BCF_T3W	3500	[mg/kg fish]/[mg/L water]	FALSE	Bioaccumulation factor (TL3 fish, whole; used for eco)	Burkhard 2021	Table 4
PFOS	BCF_T4F	1500	[mg/kg fish]/[mg/L water]	FALSE	Bioaccumulation factor (TL4 fish, filet; used for human)	Burkhard 2021	Table 4
PFOS	BCF_T4W	3500	[mg/kg fish]/[mg/L water]	FALSE	Bioaccumulation factor (TL4 fish, whole; used for eco)	Burkhard 2021	Table 4
PFOS	BrExFruit	0.03	[mg/kg DW plant]/[mg/kg soil]	FALSE	Biotransfer (soil to exposed fruit)	Blaine et al. (2014)	
PFOS	BrExVeg	1.5	[mg/kg DW plant]/[mg/kg soil]	FALSE	Biotransfer factor (soil to exposed vegetables)	Blaine et al. (2013)	
PFOS	BrForage	0.07	[mg/kg DW plant]/[mg/kg soil]	FALSE	Biotransfer factor (soil to forage)	Blaine et al. (2013)	
PFOS	BrGrain	0.07	[mg/kg DW plant]/[mg/kg soil]	FALSE	Biotransfer factor (soil to grain)	Blaine et al. (2013)	

Chemical Name	Model_Code	Value	Units	User Modified	Description	Reference	UserComment
PFOS	BrProFruit	0.03	[mg/kg DW plant]/[mg/kg soil]	FALSE	Biotransfer factor (soil to protected fruit)	Blaine et al. (2014)	
PFOS	BrProVeg	1.5	[mg/kg DW plant]/[mg/kg soil]	FALSE	Biotransfer factor (soil to protected vegetables)	Blaine et al. (2013)	
PFOS	BrSilage	0.07	[mg/kg DW plant]/[mg/kg soil]	FALSE	Biotransfer factor (soil to silage)	Blaine et al. (2013)	
PFOS	Bv	0	[ug/g DW plant]/[ug/g air]	FALSE	Biotransfer factor (vapor phase air to all plants; organics only)	No Data	No data available
PFOS	CSFOral	45.2	[per mg/kg-day]	FALSE	Oral cancer slope factor (human toxicity)	Prop PFOA-PFOS Tox	October 2022 Candidate Tox Values
PFOS	CTPWasteDry	2.15	[ug/g DW]	FALSE	Dry biosolids concentration	MI EGLE	
PFOS	Da	0.0466	[cm2/s]	FALSE	Diffusivity in air	EPA Estimation Tool	
PFOS	Dw	4.96E-6	[cm2/s]	FALSE	Diffusion coefficient in water	EPA Estimation Tool	
PFOS	Heat_of_Henry	37000	[J/mol]	FALSE	Enthalpy of phase transformation from aqueous solution to air solution	EPISuite (US EPA, 2010)	
PFOS	HLC	0	[atm-m3/mol]	FALSE	Henry's law constant	HSDB (US NLM, 2010)	Sec 6.10, HLC "<4.9E-9", set to zero due to uncertainty
PFOS	IUR	0	[ug/m3]^-1	FALSE	Inhalation unit risk (human toxicity, cancer)	No Data	
PFOS	kaer	0	(1/day)	FALSE	Aerobic biodegradation rate (surface-water column)	No Data	No data, estimation tools not appropriate
PFOS	Kanaer	0	[1/day]	FALSE	Anaerobic degradation rate (sediment)	No Data	No data, estimation tools not appropriate
PFOS	kh	0	[1/day]	FALSE	Hydrolysis rate	No Data	No data, estimation tools not appropriate
PFOS	Koc	371.5	[mL/g]	FALSE	Organic carbon partition coefficient	PFOS HESD	
PFOS	Kpo	0	(1/day)	FALSE	Photolysis degradation rate in the surface of the water column	No Data	
PFOS	ksoil	0	[1/day]	FALSE	Biodegradation rate (soil)	No Data	No data, estimation tools not appropriate
PFOS	MW	500	[g/mol]	FALSE	Molecular weight	Physprop	
PFOS	RCF	0.08	[ug/g WW plant]/[ug/mL soil water]	FALSE	Root concentration factor*	Lechner and Knapp (2011)	
PFOS	Ref_BMD_Bird	0	[mg chem/kg BW/day]	FALSE	Reference benchmark dose (bird)	No Data	
PFOS	Ref_BMD_Mammal	0	[mg chem/kg BW/day]	FALSE	Reference benchmark dose (mammal)	No Data	
PFOS	Ref_BW_Bird	0	[kg]	FALSE	Reference body weight (bird)	No Data	
PFOS	Ref_BW_Mammal	0	[kg]	FALSE	Reference body weight (mammal)	No Data	
PFOS	RFC	0	[mg/m3]	FALSE	Reference concentration (human toxicity, noncancer)	No Data	
PFOS	RfD	1.00E-07	[mg/kg-day]	FALSE	Reference dose (human toxicity, noncancer)	Prop PFOA-PFOS Tox	October 2022 Candidate Tox Values

Chemical Name	Model_Code	Value	Units	User Modified	Description	Reference	UserComment
PFOS	RFD_By_Pathway	False	NA	FALSE	True if Model run with pathway-specific RfD	System	
PFOS	Sol	370	[mg/L]	FALSE	Solubility	OECD, 2002	in fresh water
PFOS	temp_ref_aer_all	25	[C]	FALSE	Reference temperature for water column degradation	Default	
PFOS	temp_ref_anae_all	25	[C]	FALSE	Reference temperature for benthic degradation	Default	

E.2 BST Screening Results

All results are for the farm family (adult farmer or child of farmer). Pathway abbreviations are as follows:

- **Beef:** consumption of beef from beef cattle pastured on the farm
- **ExFruit:** consumption of exposed fruits grown on the farm
- **ExVeg:** consumption of exposed vegetables grown on the farm
- **Fish:** consumption of fish caught in farm pond
- **GW:** consumption of groundwater from well located on farm
- **Milk:** consumption of milk from dairy cows pastured on the farm
- **ProFruit:** consumption of protected fruits grown on the farm
- **ProVeg:** consumption of protected vegetables grown on the farm
- **Root:** consumption of root vegetables grown on the farm
- **Soil:** consumption of soil from the farm field
- **SW:** consumption of surface water from nearby reservoir.

Media concentration, dose, HQ and CRL results are presented for PFOA and PFOS in tables E-5 to E-8. In the tables with non-cancer results, the dose column represents the ADD. In the tables with cancer results the dose column represents the LADD. HQ s and CRLs for sensitive pathways, such as milk, beef, fish, and drinking water are often very high (over 100 HQ or over 1 in 1000 CRL). Note that the chicken egg consumption pathway was not included in the BST.

Table E-5. Noncancer Results: PFOA

Scenario	Receptor	Pathway	Media Conc. Units	Dry Climate			Moderate Climate			Wet Climate		
				HQ (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)	HQ (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)	HQ (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)
Crop	Adult	GW	mg/L	7,417	2.2E-04	6.5E-03	2,237	6.7E-05	2.0E-03	1,899	5.7E-05	1.7E-03
Crop	Adult	SW	mg/L	2,346	7.0E-05	2.1E-03	583	1.7E-05	5.1E-04	394	1.2E-05	3.5E-04
Crop	Adult	Fish	mg/kg WW	1,255	3.8E-05	1.4E-01	570	1.7E-05	6.1E-02	324	9.7E-06	3.5E-02
Crop	Adult	ExVeg	mg/kg WW	393	1.2E-05	2.3E-03	100	3.0E-06	5.9E-04	84	2.5E-06	5.0E-04
Crop	Adult	ProVeg	mg/kg WW	318	9.5E-06	3.0E-03	81	2.4E-06	7.7E-04	68	2.1E-06	6.5E-04
Crop	Adult	ProFruit	mg/kg WW	64	1.9E-06	1.9E-04	16	4.9E-07	4.9E-05	14	4.1E-07	4.2E-05
Crop	Adult	ExFruit	mg/kg WW	30	8.9E-07	2.3E-04	8	2.3E-07	5.7E-05	6	1.9E-07	4.8E-05
Crop	Adult	Root	mg/kg WW	0.2	6.3E-09	2.2E-06	0.05	1.6E-09	5.5E-07	0.05	1.4E-09	4.6E-07
Crop	Adult	Soil	mg/kg	0.06	1.7E-09	1.3E-02	0.009	2.8E-10	2.2E-03	0.007	2.0E-10	1.6E-03
Crop	Child	GW	mg/L	9,599	2.9E-04	6.5E-03	2,895	8.7E-05	2.0E-03	2,458	7.4E-05	1.7E-03
Crop	Child	SW	mg/L	3,036	9.1E-05	2.1E-03	754	2.3E-05	5.1E-04	510	1.5E-05	3.5E-04
Crop	Child	Fish	mg/kg WW	1,562	4.7E-05	1.4E-01	710	2.1E-05	6.1E-02	403	1.2E-05	3.5E-02
Crop	Child	ExVeg	mg/kg WW	419	1.3E-05	2.3E-03	106	3.2E-06	5.9E-04	90	2.7E-06	5.0E-04
Crop	Child	ProVeg	mg/kg WW	274	8.2E-06	3.0E-03	69	2.1E-06	7.7E-04	59	1.8E-06	6.5E-04
Crop	Child	ProFruit	mg/kg WW	73	2.2E-06	1.9E-04	19	5.6E-07	4.9E-05	16	4.7E-07	4.2E-05
Crop	Child	ExFruit	mg/kg WW	32	9.6E-07	2.3E-04	8	2.4E-07	5.7E-05	7	2.1E-07	4.8E-05
Crop	Child	Soil	mg/kg	1.2	3.5E-08	1.3E-02	0.2	6.0E-09	2.2E-03	0.14	4.2E-09	1.6E-03
Crop	Child	Root	mg/kg WW	0.4	1.2E-08	2.2E-06	0.10	3.0E-09	5.5E-07	0.08	2.5E-09	4.6E-07
Pasture	Adult	GW	mg/L	8,664	2.6E-04	7.6E-03	1,278	3.8E-05	1.1E-03	854	2.6E-05	7.5E-04
Pasture	Adult	SW	mg/L	3,366	1.0E-04	3.0E-03	878	2.6E-05	7.8E-04	587	1.8E-05	5.2E-04
Pasture	Adult	Milk	mg/kg WW	3,520	1.1E-04	3.0E-03	481	1.4E-05	4.1E-04	382	1.1E-05	3.3E-04
Pasture	Adult	Fish	mg/kg WW	1,333	4.0E-05	1.4E-01	358	1.1E-05	3.9E-02	169	5.1E-06	1.8E-02
Pasture	Adult	Beef	mg/kg WW	232	7.0E-06	2.3E-03	32	9.5E-07	3.2E-04	25	7.5E-07	2.5E-04
Pasture	Adult	Soil	mg/kg	0.2	7.1E-09	5.6E-02	0.03	9.7E-10	7.7E-03	0.02	7.4E-10	5.8E-03
Pasture	Child	GW	mg/L	11,213	3.4E-04	7.6E-03	1,654	5.0E-05	1.1E-03	1,106	3.3E-05	7.5E-04
Pasture	Child	SW	mg/L	4,356	1.3E-04	3.0E-03	1,137	3.4E-05	7.8E-04	760	2.3E-05	5.2E-04
Pasture	Child	Milk	mg/kg WW	5,934	1.8E-04	3.0E-03	812	2.4E-05	4.1E-04	643	1.9E-05	3.3E-04
Pasture	Child	Fish	mg/kg WW	1,660	5.0E-05	1.4E-01	446	1.3E-05	3.9E-02	210	6.3E-06	1.8E-02
Pasture	Child	Beef	mg/kg WW	473	1.4E-05	2.3E-03	65	1.9E-06	3.2E-04	51	1.5E-06	2.5E-04
Pasture	Child	Soil	mg/kg	5	1.5E-07	5.6E-02	0.7	2.1E-08	7.7E-03	0.5	1.6E-08	5.8E-03
Reclamation	Adult	Milk	mg/kg WW	3,726	1.1E-04	3.2E-03	3,220	9.7E-05	2.8E-03	2,483	7.4E-05	2.1E-03
Reclamation	Adult	Beef	mg/kg WW	246	7.4E-06	2.5E-03	211	6.3E-06	2.1E-03	164	4.9E-06	1.6E-03
Reclamation	Adult	GW	mg/L	139	4.2E-06	1.2E-04	163	4.9E-06	1.4E-04	112	3.4E-06	9.9E-05
Reclamation	Adult	Soil	mg/kg	0.2	7.5E-09	5.9E-02	0.2	5.8E-09	4.6E-02	0.2	4.9E-09	3.9E-02
Reclamation	Adult	SW	mg/L	0.005	1.5E-10	4.5E-09	6.6E-05	2.0E-12	5.8E-11	0.04	1.1E-09	3.3E-08
Reclamation	Adult	Fish	mg/kg WW	5.5E-04	1.65E-11	5.93E-08	6.4E-06	1.93E-13	6.92E-10	0.004	1.22E-10	4.38E-07

Scenario	Receptor	Pathway	Media Conc. Units	Dry Climate			Moderate Climate			Wet Climate		
				HQ (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)	HQ (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)	HQ (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)
Reclamation	Child	Milk	mg/kg WW	6,280	1.9E-04	3.2E-03	5,428	1.6E-04	2.8E-03	4,186	1.3E-04	2.1E-03
Reclamation	Child	Beef	mg/kg WW	500	1.5E-05	2.5E-03	430	1.3E-05	2.1E-03	333	1.0E-05	1.6E-03
Reclamation	Child	GW	mg/L	180	5.4E-06	1.2E-04	211	6.3E-06	1.4E-04	145	4.4E-06	9.9E-05
Reclamation	Child	Soil	mg/kg	5	1.6E-07	5.9E-02	4	1.2E-07	4.6E-02	3	1.0E-07	3.9E-02
Reclamation	Child	SW	mg/L	0.007	2.0E-10	4.5E-09	8.5E-05	2.5E-12	5.8E-11	0.05	1.4E-09	3.3E-08
Reclamation	Child	Fish	mg/kg WW	6.9E-04	2.1E-11	5.9E-08	8.0E-06	2.4E-13	6.9E-10	0.005	1.5E-10	4.4E-07

Table E-6. Noncancer Results: PFOS

Scenario	Receptor	Pathway	Media Conc. Units	Dry Climate			Moderate Climate			Wet Climate		
				HQ (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)	HQ (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)	HQ (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)
Crop	Adult	Fish	mg/kg WW	210,949	2.1E-02	7.6E+01	124,481	1.2E-02	4.5E+01	57,646	5.8E-03	2.1E+01
Crop	Adult	GW	mg/L	9,281	9.3E-04	2.7E-02	2,805	2.8E-04	8.3E-03	2,314	2.3E-04	6.8E-03
Crop	Adult	SW	mg/L	3,067	3.1E-04	9.0E-03	1,499	1.5E-04	4.4E-03	993	9.9E-05	2.9E-03
Crop	Adult	ExVeg	mg/kg WW	1,171	1.2E-04	2.3E-02	389	3.9E-05	7.7E-03	301	3.0E-05	6.0E-03
Crop	Adult	ProVeg	mg/kg WW	948	9.5E-05	3.0E-02	315	3.2E-05	1.0E-02	243	2.4E-05	7.8E-03
Crop	Adult	ProFruit	mg/kg WW	56	5.6E-06	5.6E-04	18	1.8E-06	1.9E-04	14	1.4E-06	1.4E-04
Crop	Adult	ExFruit	mg/kg WW	26	2.6E-06	6.5E-04	9	8.6E-07	2.2E-04	7	6.6E-07	1.7E-04
Crop	Adult	Root	mg/kg WW	0.8	8.3E-08	2.8E-05	0.3	2.8E-08	9.4E-06	0.2	2.1E-08	7.3E-06
Crop	Adult	Soil	mg/kg	0.2	1.8E-08	1.4E-01	0.05	4.8E-09	3.8E-02	0.03	2.9E-09	2.3E-02
Crop	Child	Fish	mg/kg WW	262,600	2.6E-02	7.6E+01	154,960	1.5E-02	4.5E+01	71,760	7.2E-03	2.1E+01
Crop	Child	GW	mg/L	12,011	1.2E-03	2.7E-02	3,630	3.6E-04	8.3E-03	2,995	3.0E-04	6.8E-03
Crop	Child	SW	mg/L	3,969	4.0E-04	9.0E-03	1,940	1.9E-04	4.4E-03	1,285	1.3E-04	2.9E-03
Crop	Child	ExVeg	mg/kg WW	1,250	1.2E-04	2.3E-02	415	4.2E-05	7.7E-03	321	3.2E-05	6.0E-03
Crop	Child	ProVeg	mg/kg WW	816	8.2E-05	3.0E-02	271	2.7E-05	1.0E-02	209	2.1E-05	7.8E-03
Crop	Child	ProFruit	mg/kg WW	63	6.3E-06	5.6E-04	21	2.1E-06	1.9E-04	16	1.6E-06	1.4E-04
Crop	Child	ExFruit	mg/kg WW	28	2.8E-06	6.5E-04	9	9.2E-07	2.2E-04	7	7.1E-07	1.7E-04
Crop	Child	Soil	mg/kg	4	3.8E-07	1.4E-01	1.0	1.0E-07	3.8E-02	0.6	6.2E-08	2.3E-02
Crop	Child	Root	mg/kg WW	2	1.5E-07	2.8E-05	0.5	5.1E-08	9.4E-06	0.4	3.9E-08	7.3E-06
Pasture	Adult	Fish	mg/kg WW	279,038	2.8E-02	1.0E+02	58,899	5.9E-03	2.1E+01	26,066	2.6E-03	9.4E+00
Pasture	Adult	GW	mg/L	4,068	4.1E-04	1.2E-02	2,186	2.2E-04	6.4E-03	1,561	1.6E-04	4.6E-03
Pasture	Adult	SW	mg/L	5,882	5.9E-04	1.7E-02	1,734	1.7E-04	5.1E-03	1,017	1.0E-04	3.0E-03
Pasture	Adult	Milk	mg/kg WW	10,898	1.1E-03	3.1E-02	1,255	1.3E-04	3.6E-03	1,088	1.1E-04	3.1E-03
Pasture	Adult	Beef	mg/kg WW	2,451	2.5E-04	8.2E-02	281	2.8E-05	9.4E-03	242	2.4E-05	8.1E-03
Pasture	Adult	Soil	mg/kg	1.1	1.1E-07	8.9E-01	0.13	1.3E-08	9.9E-02	0.11	1.1E-08	8.4E-02
Pasture	Child	Fish	mg/kg WW	347,360	3.5E-02	1.0E+02	73,320	7.3E-03	2.1E+01	32,448	3.2E-03	9.4E+00
Pasture	Child	GW	mg/L	5,264	5.3E-04	1.2E-02	2,829	2.8E-04	6.4E-03	2,021	2.0E-04	4.6E-03

Scenario	Receptor	Pathway	Media Conc. Units	Dry Climate			Moderate Climate			Wet Climate		
				HQ (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)	HQ (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)	HQ (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)
Pasture	Child	SW	mg/L	7,612	7.6E-04	1.7E-02	2,244	2.2E-04	5.1E-03	1,316	1.3E-04	3.0E-03
Pasture	Child	Milk	mg/kg WW	18,371	1.8E-03	3.1E-02	2,115	2.1E-04	3.6E-03	1,833	1.8E-04	3.1E-03
Pasture	Child	Beef	mg/kg WW	4,992	5.0E-04	8.2E-02	572	5.7E-05	9.4E-03	494	4.9E-05	8.1E-03
Pasture	Child	Soil	mg/kg	24	2.4E-06	8.9E-01	3	2.6E-07	9.9E-02	2	2.2E-07	8.4E-02
Reclamation	Adult	Milk	mg/kg WW	5,237	5.2E-04	1.5E-02	4,864	4.9E-04	1.4E-02	3,828	3.8E-04	1.1E-02
Reclamation	Adult	Beef	mg/kg WW	1,178	1.2E-04	3.9E-02	1,086	1.1E-04	3.6E-02	846	8.5E-05	2.8E-02
Reclamation	Adult	GW	mg/L	167	1.7E-05	4.9E-04	83	8.3E-06	2.4E-04	54	5.4E-06	1.6E-04
Reclamation	Adult	Soil	mg/kg	0.5	5.4E-08	4.3E-01	0.5	4.8E-08	3.8E-01	0.4	3.8E-08	3.0E-01
Reclamation	Adult	Fish	mg/kg WW	0.2	1.5E-08	5.4E-05	0.002	1.8E-10	6.4E-07	1.2	1.2E-07	4.4E-04
Reclamation	Adult	SW	mg/L	0.009	8.6E-10	2.5E-08	1.1E-04	1.1E-11	3.2E-10	0.07	6.8E-09	2.0E-07
Reclamation	Child	Milk	mg/kg WW	8,829	8.8E-04	1.5E-02	8,199	8.2E-04	1.4E-02	6,453	6.5E-04	1.1E-02
Reclamation	Child	Beef	mg/kg WW	2,400	2.4E-04	3.9E-02	2,213	2.2E-04	3.6E-02	1,723	1.7E-04	2.8E-02
Reclamation	Child	GW	mg/L	217	2.2E-05	4.9E-04	107	1.1E-05	2.4E-04	70	7.0E-06	1.6E-04
Reclamation	Child	Soil	mg/kg	11	1.1E-06	4.3E-01	10	1.0E-06	3.8E-01	8	8.0E-07	3.0E-01
Reclamation	Child	Fish	mg/kg WW	0.2	1.9E-08	5.4E-05	0.002	2.2E-10	6.4E-07	2	1.5E-07	4.4E-04
Reclamation	Child	SW	mg/L	0.011	1.1E-09	2.5E-08	1.4E-04	1.4E-11	3.2E-10	0.09	8.8E-09	2.0E-07

Table E-7. Cancer Results: PFOA

Scenario	Receptor	Pathway	Media Conc. Units	Dry Climate			Moderate Climate			Wet Climate		
				Risk (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)	Risk (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)	Risk (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)
Crop	Adult	GW	mg/L	4.2E+00	1.4E-04	6.4E-03	9.4E-01	3.2E-05	1.4E-03	9.2E-01	3.2E-05	1.4E-03
Crop	Adult	SW	mg/L	1.0E+00	3.4E-05	1.5E-03	2.4E-01	8.3E-06	3.7E-04	1.6E-01	5.3E-06	2.4E-04
Crop	Adult	Fish	mg/kg WW	5.4E-01	1.8E-05	1.0E-01	2.3E-01	7.8E-06	4.2E-02	1.3E-01	4.3E-06	2.4E-02
Crop	Adult	ExVeg	mg/kg WW	1.9E-01	6.5E-06	2.0E-03	4.1E-02	1.4E-06	4.2E-04	3.5E-02	1.2E-06	3.6E-04
Crop	Adult	ProVeg	mg/kg WW	1.5E-01	5.3E-06	2.6E-03	3.3E-02	1.1E-06	5.5E-04	2.9E-02	9.7E-07	4.7E-04
Crop	Adult	ProFruit	mg/kg WW	3.1E-02	1.1E-06	1.6E-04	6.7E-03	2.3E-07	3.5E-05	5.8E-03	2.0E-07	3.0E-05
Crop	Adult	ExFruit	mg/kg WW	1.4E-02	4.9E-07	1.9E-04	3.1E-03	1.1E-07	4.1E-05	2.7E-03	9.1E-08	3.5E-05
Crop	Adult	Root	mg/kg WW	1.0E-04	3.5E-09	1.8E-06	2.2E-05	7.5E-10	3.9E-07	1.9E-05	6.5E-10	3.4E-07
Crop	Adult	Soil	mg/kg	1.5E-05	5.1E-10	6.2E-03	2.1E-06	7.2E-11	8.7E-04	1.6E-06	5.3E-11	6.4E-04
Crop	Child	GW	mg/L	1.2E+00	4.0E-05	6.4E-03	3.6E-01	1.2E-05	2.0E-03	3.1E-01	1.0E-05	1.7E-03
Crop	Child	SW	mg/L	3.3E-01	1.1E-05	1.9E-03	8.6E-02	2.9E-06	4.7E-04	5.4E-02	1.8E-06	3.0E-04
Crop	Child	Fish	mg/kg WW	1.7E-01	5.9E-06	1.2E-01	7.2E-02	2.4E-06	5.0E-02	4.0E-02	1.4E-06	2.8E-02
Crop	Child	ExVeg	mg/kg WW	4.3E-02	1.5E-06	2.3E-03	9.9E-03	3.4E-07	5.3E-04	7.9E-03	2.7E-07	4.2E-04
Crop	Child	ProVeg	mg/kg WW	3.3E-02	1.1E-06	2.9E-03	7.6E-03	2.6E-07	6.9E-04	6.1E-03	2.1E-07	5.5E-04
Crop	Child	ProFruit	mg/kg WW	1.0E-02	3.5E-07	1.9E-04	2.4E-03	8.2E-08	4.4E-05	1.9E-03	6.5E-08	3.5E-05
Crop	Child	ExFruit	mg/kg WW	5.2E-03	1.8E-07	2.2E-04	1.2E-03	4.2E-08	5.1E-05	9.8E-04	3.4E-08	4.1E-05

Scenario	Receptor	Pathway	Media Conc. Units	Dry Climate			Moderate Climate			Wet Climate		
				Risk (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)	Risk (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)	Risk (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)
Crop	Child	Root	mg/kg WW	4.5E-05	1.5E-09	2.1E-06	1.0E-05	3.5E-10	4.9E-07	8.2E-06	2.8E-10	3.9E-07
Crop	Child	Soil	mg/kg	8.2E-05	2.8E-09	8.7E-03	9.3E-06	3.2E-10	1.2E-03	5.9E-06	2.0E-10	8.2E-04
Pasture	Adult	GW	mg/L	2.8E+00	9.5E-05	4.3E-03	5.7E-01	1.9E-05	8.7E-04	3.2E-01	1.1E-05	4.9E-04
Pasture	Adult	SW	mg/L	1.6E+00	5.4E-05	2.4E-03	3.7E-01	1.2E-05	5.6E-04	2.3E-01	8.0E-06	3.6E-04
Pasture	Adult	Fish	mg/kg WW	6.1E-01	2.1E-05	1.1E-01	1.5E-01	5.1E-06	2.8E-02	6.3E-02	2.2E-06	1.2E-02
Pasture	Adult	Milk	mg/kg WW	9.1E-01	3.1E-05	1.3E-03	1.1E-01	3.9E-06	1.7E-04	7.5E-02	2.5E-06	1.1E-04
Pasture	Adult	Beef	mg/kg WW	6.0E-02	2.0E-06	1.0E-03	7.5E-03	2.6E-07	1.3E-04	4.9E-03	1.7E-07	8.5E-05
Pasture	Adult	Soil	mg/kg	6.0E-05	2.0E-09	2.4E-02	7.1E-06	2.4E-10	2.9E-03	4.6E-06	1.6E-10	1.9E-03
Pasture	Child	GW	mg/L	9.3E-01	3.2E-05	4.5E-03	2.1E-01	7.0E-06	1.1E-03	1.4E-01	4.7E-06	7.5E-04
Pasture	Child	SW	mg/L	5.0E-01	1.7E-05	2.8E-03	1.2E-01	4.1E-06	6.7E-04	8.7E-02	3.0E-06	4.8E-04
Pasture	Child	Fish	mg/kg WW	2.0E-01	6.7E-06	1.3E-01	4.8E-02	1.6E-06	3.3E-02	2.4E-02	8.1E-07	1.6E-02
Pasture	Child	Milk	mg/kg WW	4.1E-01	1.4E-05	1.9E-03	4.1E-02	1.4E-06	2.2E-04	3.2E-02	1.1E-06	1.7E-04
Pasture	Child	Beef	mg/kg WW	4.6E-02	1.6E-06	1.5E-03	5.0E-03	1.7E-07	1.7E-04	3.8E-03	1.3E-07	1.3E-04
Pasture	Child	Soil	mg/kg	3.2E-04	1.1E-08	3.5E-02	3.0E-05	1.0E-09	3.9E-03	2.3E-05	8.0E-10	2.8E-03
Reclamation	Adult	Milk	mg/kg WW	5.3E-01	1.8E-05	7.8E-04	9.0E-02	3.1E-06	1.3E-04	5.9E-02	2.0E-06	8.8E-05
Reclamation	Adult	GW	mg/L	8.0E-02	2.7E-06	1.2E-04	7.6E-02	2.6E-06	1.2E-04	2.7E-02	9.2E-07	4.1E-05
Reclamation	Adult	Beef	mg/kg WW	3.4E-02	1.2E-06	6.0E-04	5.9E-03	2.0E-07	1.0E-04	3.9E-03	1.3E-07	6.7E-05
Reclamation	Adult	Soil	mg/kg	3.3E-05	1.1E-09	1.4E-02	5.2E-06	1.8E-10	2.1E-03	3.4E-06	1.2E-10	1.4E-03
Reclamation	Adult	SW	mg/L	2.9E-06	1.0E-10	4.5E-09	3.2E-08	1.1E-12	4.9E-11	8.9E-06	3.0E-10	1.4E-08
Reclamation	Adult	Fish	mg/kg WW	3.2E-07	1.1E-11	5.9E-08	3.1E-09	1.1E-13	5.9E-10	9.7E-07	3.3E-11	1.8E-07
Reclamation	Child	Milk	mg/kg WW	4.6E-01	1.6E-05	2.1E-03	1.5E-01	5.2E-06	4.9E-04	1.0E-01	3.4E-06	3.2E-04
Reclamation	Child	GW	mg/L	2.2E-02	7.7E-07	1.2E-04	2.6E-02	9.0E-07	1.4E-04	1.8E-02	6.2E-07	9.9E-05
Reclamation	Child	Beef	mg/kg WW	4.8E-02	1.7E-06	1.6E-03	1.2E-02	4.1E-07	3.8E-04	7.9E-03	2.7E-07	2.5E-04
Reclamation	Child	Soil	mg/kg	3.4E-04	1.2E-08	3.7E-02	1.1E-04	3.8E-09	7.9E-03	7.2E-05	2.5E-09	5.2E-03
Reclamation	Child	SW	mg/L	8.1E-07	2.7E-11	4.4E-09	1.0E-08	3.5E-13	5.7E-11	5.9E-06	2.0E-10	3.2E-08
Reclamation	Child	Fish	mg/kg WW	8.5E-08	2.9E-12	5.8E-08	9.9E-10	3.4E-14	6.8E-10	6.3E-07	2.1E-11	4.3E-07

Table E-8. Cancer Results: PFOS

Scenario	Receptor	Pathway	Media Conc. Units	Dry Climate			Moderate Climate			Wet Climate		
				Risk (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)	Risk (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)	Risk (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)
Crop	Adult	Fish	mg/kg WW	4.5E-01	1.0E-02	5.4E+01	2.4E-01	5.4E-03	2.9E+01	1.2E-01	2.7E-03	1.5E+01
Crop	Adult	GW	mg/L	2.8E-02	6.1E-04	2.7E-02	6.4E-03	1.4E-04	6.3E-03	5.9E-03	1.3E-04	5.8E-03
Crop	Adult	SW	mg/L	6.6E-03	1.5E-04	6.5E-03	3.0E-03	6.7E-05	3.0E-03	2.0E-03	4.4E-05	2.0E-03
Crop	Adult	ExVeg	mg/kg WW	2.9E-03	6.4E-05	1.9E-02	9.2E-04	2.0E-05	6.1E-03	7.1E-04	1.6E-05	4.8E-03
Crop	Adult	ProVeg	mg/kg WW	2.3E-03	5.2E-05	2.5E-02	7.4E-04	1.6E-05	8.0E-03	5.8E-04	1.3E-05	6.2E-03
Crop	Adult	ProFruit	mg/kg WW	1.4E-04	3.0E-06	4.6E-04	4.4E-05	9.6E-07	1.5E-04	3.4E-05	7.5E-07	1.1E-04

Scenario	Receptor	Pathway	Media Conc. Units	Dry Climate			Moderate Climate			Wet Climate		
				Risk (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)	Risk (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)	Risk (unitless)	Dose (mg/kg-d)	Media Conc. (units see left)
Crop	Adult	ExFruit	mg/kg WW	6.3E-05	1.4E-06	5.4E-04	2.0E-05	4.5E-07	1.7E-04	1.6E-05	3.5E-07	1.3E-04
Crop	Adult	Root	mg/kg WW	2.0E-06	4.5E-08	2.3E-05	6.5E-07	1.4E-08	7.5E-06	5.1E-07	1.1E-08	5.8E-06
Crop	Adult	Soil	mg/kg	2.5E-07	5.6E-09	6.8E-02	5.9E-08	1.3E-09	1.6E-02	4.3E-08	9.5E-10	1.1E-02
Crop	Child	Fish	mg/kg WW	1.5E-01	3.3E-03	6.8E+01	8.4E-02	1.9E-03	3.8E+01	4.2E-02	9.2E-04	1.9E+01
Crop	Child	GW	mg/L	7.7E-03	1.7E-04	2.7E-02	2.3E-03	5.2E-05	8.3E-03	1.9E-03	4.3E-05	6.8E-03
Crop	Child	SW	mg/L	2.2E-03	4.8E-05	7.9E-03	1.1E-03	2.4E-05	3.9E-03	6.8E-04	1.5E-05	2.5E-03
Crop	Child	ExVeg	mg/kg WW	6.4E-04	1.4E-05	2.2E-02	2.1E-04	4.7E-06	7.3E-03	1.6E-04	3.5E-06	5.5E-03
Crop	Child	ProVeg	mg/kg WW	4.9E-04	1.1E-05	2.9E-02	1.6E-04	3.6E-06	9.5E-03	1.2E-04	2.7E-06	7.2E-03
Crop	Child	ProFruit	mg/kg WW	4.5E-05	9.9E-07	5.3E-04	1.5E-05	3.3E-07	1.8E-04	1.1E-05	2.5E-07	1.3E-04
Crop	Child	ExFruit	mg/kg WW	2.3E-05	5.1E-07	6.2E-04	7.6E-06	1.7E-07	2.0E-04	5.8E-06	1.3E-07	1.6E-04
Crop	Child	Root	mg/kg WW	8.8E-07	2.0E-08	2.7E-05	2.9E-07	6.4E-09	8.9E-06	2.2E-07	4.9E-09	6.7E-06
Crop	Child	Soil	mg/kg	1.4E-06	3.1E-08	1.0E-01	2.7E-07	6.0E-09	2.3E-02	1.9E-07	4.1E-09	1.6E-02
Pasture	Adult	Fish	mg/kg WW	6.0E-01	1.3E-02	7.3E+01	1.3E-01	2.9E-03	1.6E+01	5.7E-02	1.3E-03	6.8E+00
Pasture	Adult	GW	mg/L	1.2E-02	2.7E-04	1.2E-02	5.5E-03	1.2E-04	5.4E-03	3.9E-03	8.6E-05	3.8E-03
Pasture	Adult	SW	mg/L	1.3E-02	3.0E-04	1.3E-02	3.5E-03	7.8E-05	3.5E-03	2.3E-03	5.0E-05	2.2E-03
Pasture	Adult	Milk	mg/kg WW	2.0E-02	4.3E-04	1.9E-02	2.6E-03	5.7E-05	2.5E-03	1.7E-03	3.8E-05	1.7E-03
Pasture	Adult	Beef	mg/kg WW	4.4E-03	9.7E-05	4.9E-02	5.8E-04	1.3E-05	6.5E-03	3.9E-04	8.6E-06	4.3E-03
Pasture	Adult	Soil	mg/kg	2.0E-06	4.4E-08	5.3E-01	2.6E-07	5.7E-09	6.8E-02	1.7E-07	3.8E-09	4.5E-02
Pasture	Child	Fish	mg/kg WW	2.0E-01	4.5E-03	9.1E+01	4.2E-02	9.4E-04	1.9E+01	1.9E-02	4.1E-04	8.3E+00
Pasture	Child	GW	mg/L	3.4E-03	7.5E-05	1.2E-02	1.8E-03	4.0E-05	6.4E-03	1.3E-03	2.9E-05	4.6E-03
Pasture	Child	SW	mg/L	4.2E-03	9.3E-05	1.5E-02	1.1E-03	2.4E-05	3.9E-03	6.2E-04	1.4E-05	2.3E-03
Pasture	Child	Milk	mg/kg WW	8.5E-03	1.9E-04	2.7E-02	8.4E-04	1.9E-05	2.9E-03	5.9E-04	1.3E-05	2.0E-03
Pasture	Child	Beef	mg/kg WW	3.2E-03	7.2E-05	7.2E-02	3.3E-04	7.3E-06	7.5E-03	2.4E-04	5.3E-06	5.2E-03
Pasture	Child	Soil	mg/kg	1.0E-05	2.3E-07	7.7E-01	9.7E-07	2.1E-08	7.9E-02	6.8E-07	1.5E-08	5.5E-02
Reclamation	Adult	Milk	mg/kg WW	8.0E-03	1.8E-04	7.6E-03	1.3E-03	2.9E-05	1.2E-03	1.0E-03	2.3E-05	9.9E-04
Reclamation	Adult	Beef	mg/kg WW	1.8E-03	3.9E-05	2.0E-02	2.8E-04	6.3E-06	3.2E-03	2.2E-04	5.0E-06	2.5E-03
Reclamation	Adult	GW	mg/L	5.0E-04	1.1E-05	4.9E-04	2.5E-04	5.4E-06	2.4E-04	1.6E-04	3.6E-06	1.6E-04
Reclamation	Adult	Soil	mg/kg	7.9E-07	1.7E-08	2.1E-01	1.1E-07	2.5E-09	3.0E-02	8.8E-08	1.9E-09	2.3E-02
Reclamation	Adult	Fish	mg/kg WW	4.5E-07	9.9E-09	5.4E-05	5.2E-09	1.2E-10	6.3E-07	3.6E-06	8.0E-08	4.4E-04
Reclamation	Adult	SW	mg/L	2.5E-08	5.6E-10	2.5E-08	3.2E-10	7.2E-12	3.2E-10	2.0E-07	4.5E-09	2.0E-07
Reclamation	Child	Milk	mg/kg WW	3.5E-03	7.8E-05	1.1E-02	2.0E-03	4.3E-05	4.5E-03	1.6E-03	3.6E-05	3.6E-03
Reclamation	Child	Beef	mg/kg WW	1.3E-03	2.9E-05	2.9E-02	5.7E-04	1.3E-05	1.2E-02	4.5E-04	1.0E-05	9.3E-03
Reclamation	Child	GW	mg/L	1.4E-04	3.1E-06	4.9E-04	6.9E-05	1.5E-06	2.4E-04	4.5E-05	9.9E-07	1.6E-04
Reclamation	Child	Soil	mg/kg	4.1E-06	9.0E-08	3.1E-01	2.2E-06	4.8E-08	1.1E-01	1.8E-06	4.0E-08	8.6E-02
Reclamation	Child	Fish	mg/kg WW	1.2E-07	2.7E-09	5.4E-05	1.4E-09	3.1E-11	6.3E-07	9.7E-07	2.1E-08	4.3E-04
Reclamation	Child	SW	mg/L	7.1E-09	1.6E-10	2.5E-08	9.0E-11	2.0E-12	3.2E-10	5.6E-08	1.2E-09	2.0E-07

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